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Proceedings: Sagebrush Steppe Ecosystems Symposium

Boise State University
Boise, Idaho
June 21-23, 1999

Including:

Summary Report: Snake River Birds of Prey National Conservation Area Habitat Restoration Workshop

Boise State University
Boise, Idaho
June 23-25, 1999

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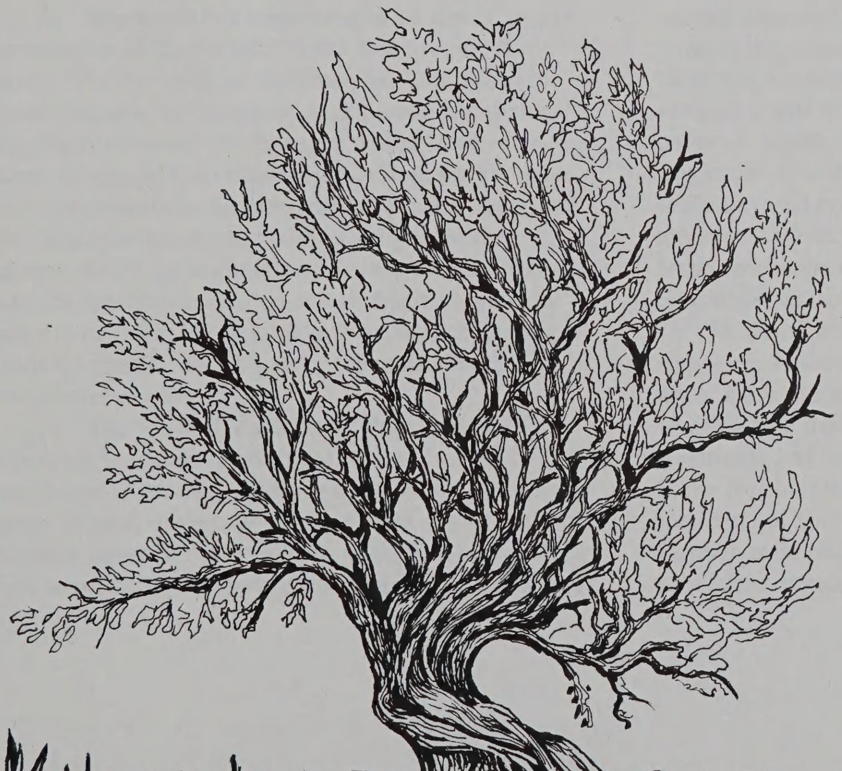

INTRODUCTORY REMARKS

J. David Brunner

Welcome! For the next two days we will discuss issues surrounding the ecology, uses, and management of sagebrush steppe ecosystems in the western United States, with emphasis on the Great Basin Desert and Columbia Plateau. The goal of this symposium is to identify practical – and I emphasize practical – solutions to stem the tide of loss and improve our abilities to restore sagebrush ecosystems.

BLM has a real interest in the presentations and the knowledge disseminated at this symposium for several reasons:

1) BLM is the largest manager of sagebrush ecosystems in this country and perhaps in the world. Many of our resource values and uses (grazing, watershed function [e.g. clean water], and recreation) are associated with sagebrush rangelands.

2) Sagebrush is a keystone species that is, in part, an indicator of the "health" of the entire region it inhabits. For example, as sagebrush has diminished in cover and area, we have seen sage grouse populations steadily decline until its listing as a threatened or endangered species is now imminent. A local weekly paper in Boise recently commented on this decline and asked the rhetorical question, "Is the sage grouse the next spotted owl?" As land managers, we would all like to turn the sage grouse and sagebrush decline around in order to maintain the flexibility to manage these rangelands for "health" as well as for the multiple uses that our publics expect.

3) The Snake River Birds of Prey National Conservation Area south of Boise is the home of the largest population of nesting raptors in North America. Loss of shrub habitat, especially sagebrush, is one of the most pressing issues in this important wildlife habitat area. In fact, immediately following this symposium, a group of scientists, managers, and land users will meet for 2 1/2 days to begin developing a strategy to reduce the loss of shrub habitat and restore areas now dominated by cheatgrass, an exotic, highly flammable annual grass.

4) Another reason we must maintain or improve the conditions of our sagebrush rangelands is found in the Idaho Standards for Rangeland Health and Guidelines for Livestock Grazing Management published in August 1997. These Standards and Guidelines, developed by our three public "Resource Advisory Councils" in Idaho, direct our management to restore or maintain "healthy, productive, and diverse native animal habitat and populations of native plants" by implementing proper grazing management practices on our public lands.

5) Finally, noxious weeds are in the forefront of our management today because of their potential to degrade or dominate disturbed sagebrush steppe rangelands. However, we have observed that our drier big sagebrush sites can resist invasion by rush skeletonweed if a good cover of sagebrush is maintained on the site. Once sagebrush is lost through various disturbances, rush skeletonweed is much more apt to invade and eventually dominate these sites. Again, a healthy, intact sagebrush landscape is more resistant to the invasion of at least some noxious weeds.

These are only a few of the reasons why sagebrush steppe ecosystems are valued; we will hear a lot more about them in the next couple of days. I would like to close by first thanking Boise State University for co-sponsoring and hosting this symposium. Our thanks also to the Northwest Chapter of the Society for Ecological Restoration and the USGS's Forest and Rangeland Ecosystem Science Center for their interest and sponsorship of the symposium.

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SCIENCE, POLITICS AND ECOSYSTEMS: THOUGHTS ON THEIR INTEGRATION

John Freemuth

The move toward an ecosystem-based approach to the management of our public lands must overcome two fundamental problems. One problem could be called the problem of science; the other, the problem of politics. The two problems are related, as perhaps this premise of mine illustrates: science is a necessary but insufficient condition for public decision making.

Let us start with the problem of science. It has certainly become clear that we cannot make effective rangeland policy without solid scientific information – often the laws require it. As a member of the BLM Science Advisory Board, I can tell you that one of our key tasks is figuring out how to get science to the managers who need it most and understanding barriers to the use of science in that bureau. Science can be seen as a problem for a number of reasons. One, there is some confusion about which science should be followed. Looking at our National Forests for a moment, it is equally valid to apply the science of forestry or the science of ecology to pressing management and policy issues. These sciences offer different perspectives, and it is often because they are underpinned by different values. Forestry developed in part with a perspective that looked at forests as tree farms, as places to be wisely managed for the good of society – in this case, for the production of goods and services thought to have economic benefit for large numbers of people. Ecology, on the other hand, tends to look at forests more as "mother earth ," as places to be protected from the ravages of industrial society. Thus, any statements regarding the use of the best science to guide decision makers are rendered problematic at best once we understand the value choices that often lie behind the use of science. Elizabeth Bird put it well when she reminded us:

> *Should we believe everything the science of ecology has to tell us about our relations with nature? Or should we examine the social construction of ecology itself ... and find out if we would want the kind of world that ecology would construct for us if it were to win political hegemony in the sciences?*

Mother earth trumps tree farms, as it were.

Closely tied to this observation is the growing use of what I term "advocacy science." Advocacy science can take two closely related forms. The first clearly mixes up values and science, where what is a clear value preference ends up masked as a scientific truth. The second works by adopting a certain value preference as a policy goal (logging is harmful) and then attempts to "find the science" that demands a certain conclusion that turns out to be the pre-chosen goal (science tells us that logging harms biodiversity, therefore we must stop logging).

Consider the following example: In the December 1994 issue of Conservation Biology, a fascinating editorial was written about the role of conservation biology in range management questions. The opinion piece takes issue with a question asked by Reed Noss, which is whether conservation biologists should "link arms with activists in efforts to reform grazing practices." The authors' conclusions are negative. Worried that conservation biologists would damage their credibility by openly advocating political positions, the authors instead suggest asking a different question: "How can livestock grazing be managed to have the fewest impacts on biodiversity and ecosystem integrity?" The authors claim that a special journal symposium on grazing which precipitated their editorial offered no help on this question. Then, in a powerful conclusion to their editorial, we read:

> *The inherent flaw of deductive reasoning asks one simply to accept that "range management must be dramatically reformed." How could we continue to conduct this research and attempt to develop valid results if we worked from that premise? Our work as scientists involves recognizing patterns based on data and only then formulating a general rule. More importantly, how can we hope to advance the Society's mission to preserve biological diversity if our audience of policymakers assumes that we intend to "prove" a presumed conclusion instead of attempting to falsify well-framed null hypotheses?*

Finally, public trust in expertise – at least expertise in a general sense – has declined. In our own area of natural resources, the public wonders out loud when it is told that fire is good for the ecosystem after having been told for years by similar people that "only you can prevent forest fires." They are buffeted by a myriad of

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talking heads that talk endlessly to each other about this or that policy topic. Is it any wonder folks turn off their TVs in disgust, convinced that everything causes cancer and that their views are essentially irrelevant to the greatest experts of the day?

The Problem of Politics

Politics present a different set of problems and issues, which must be understood in order to better manage and protect ecosystems. First, the U.S. political system is designed to check and fragment power; hence, moving in the direction of ecosystem protection takes a good deal of time and effort. Those who advocate for ecosystem protection need to be fully aware of how our current institutional arrangements affect the success of implementing ecosystem protection as a management paradigm. Note, though, that these arrangements are based on assumptions that lead to structuring of political power relationships in a certain way.

There is no better voice here than that of James Madison, who explains one of the key assumptions of the authors of the Constitution this way:

> *Ambition must be made to counteract ambition.... If men were angels, no government would be necessary. If angels were to govern men, neither external nor internal controls on government would be necessary. In framing a government of men over men, the great difficulty lies in this: you must first enable the government to control the governed and in the next pace, oblige it to control itself. A dependence on the people is, no doubt, the primary control on the government, but experience has taught mankind the necessity of auxiliary precautions.*

The precautions, of course, are the commonly understood checks and balances, separation of powers, federalism, and republicanism. Power is diffused in the U.S. political system. Policy change is often difficult to achieve.

Madison, in Federalist 10, notes that one of the most important reasons for checking power is the existence of factions (today we would call them interest groups). A faction is "a majority or minority of the whole who are united and actuated by some common impulse of passion, or of interest, adverse to the rights of other citizens or to the permanent and aggregate interests of the community." Hence, the need to check Madison's "mischiefs of faction" by representative government, larger political units, and so forth.

Putting all of the above in more modern terms, there is, thus, a designed tendency of the political system to gridlock and for policy shifts to happen rarely. But we do know that we have seen instances where our political system overcame the tendency for political gridlock. One example of particular interest to proponents of ecosystem management is the development of certain policies during the Progressive Era at the turn of the last century.

Practitioners interested in the implementation of an ecosystem-based management regime would do well to revisit the early days of the Progressive Movement for clues as to how to develop and implement a management regime accepted by an entire society. We remember this era as the time of Gifford Pinchot, Teddy Roosevelt, and the birth of the Conservation Movement. The Progressive Era, of course, institutionalized science-based, expert-centered management as a general approach to the growing complexity of society at the time. For example, the federal bureau that best represented the Progressive Era in land management was the United States Forest Service. Samuel Hays, in his seminal work Conservation and the Gospel of Efficiency, noted that:

> *Conservationists were led by people who promoted the "rational" use of resources, with a focus on efficiency, planning for future use, and the application of expertise to broad national problems. But they also promoted a system of decision making consistent with that spirit, a process by which the expert would decide in terms of the most efficient dovetailing of all competing resource users according to criteria which were considered to be objective, rational, and above the give-and-take of political conflict.*

In the case of the Forest Service, for example, the expertise brought to bear on forest management questions came from the science of forestry.

What is most important about that earlier movement, however, may well be how its themes captured the public imagination. Advocates, as well as students of ecosystem management, should pay close attention to that earlier time. Gifford Pinchot discovered that "in the long run, forestry cannot succeed unless the people who live in and near the forest are for it and not against it." Pinchot helped lead the effort for professional management of the National Forests. But the key to Pinchot's success lay not solely in his advocacy of professionalism and expertise, but in the service of both to a democratic vision.

In the words of Bob Pepperman Taylor, "For Pinchot, the conservation of natural resources is of fundamental democratic value because it allows for the possibility of equality of opportunity (access to public resources) for all citizens." Taylor adds, "If we remove the vision of Progressive democracy from Pinchot's work, we are left merely with the scientific management and control of nature for no other purpose than brute human survival."

It is also true that later foresters, as noted by David Clary, "became progressively more narrow in outlook as a result of the kind of specialized education they (Pinchot) encouraged." The vision may have become less successful over time because it lost its ability to speak in nonspecialized terms. The point to remember, though, is that early public land management was successful because of its link to a democratic vision accepted by the majority of society at the time, representing an underlying consensus about how a large amount, but not all, of our federal estate should be managed.

The above, however, can be viewed, perhaps, as a road map for the eventual integration of today's science and politics. Today there are a number of newer complications that need consideration as well. The first of those is the increasing use of political appointees at lower levels in the public bureaucracies to move bureau policy in directions sought after by Presidents and other senior officials. The term for this phenomenon is the administrative presidency. Presidents since Richard Nixon have practiced the strategy. Under this strategy, bureaus can be subject to policy shifts from administration to administration, which vary greatly and can cause undue stress on professionals within bureaus.

A second complication concerns the push toward collaborative decision making. What remains unresolved is the role of national versus local groups in terms of representation at the collaborative table. The problem is whether national interests have taken the place of local values, say, in the case of local and national environmental groups. Environmental values may be represented through local groups, but clearly the national groups have their own interests which often lead them to oppose local decision making, even when environmental values are well represented.

Third, internal bureau organization presents interesting political issues too. Many federal bureaus have dominant professions within them that make up the desired path toward line management positions within the bureau. Any move toward ecosystem management must take into account the sort of management skills needed for the collaborative, cross-jurisdictional approach demanded. The issue should not be whether degrees in ecology (as, say, forestry before) should dominate the line positions but, rather, what skills make for a good ecosystem manager.

Fourth, we must pay close attention to the definition of the problem we are trying to solve. There is no correct way to define a problem, and defining a problem is a political act. Note how, in the symposium program "Welcome," we read about the negative effects of "human encroachment." This is probably true from an observational point of view but also suggests that human encroachment should be curtailed if not reversed. Such a blanket assertion may lead to a good deal of opposition from those who perceive that this will lead to more restrictions on human activity in the name of ecosystem protection.

What is the prescription then? I would suggest that those involved in research, management, and protection of sagebrush ecosystems lay out their vision of why our sagebrush steppe ecosystems are worth our protection. But expect to have an active and involved conversation with those who would like to know more or are in opposition with suggested protection policies that might develop. Science can inform this conversation, but it alone cannot arrive at enforceable goals and purposes for those desert ecosystems. As Wallace Stegner once reminded us: a place is nothing in itself. It has no meaning; it can hardly be said to exist except in terms of human perception, use, and response.

Historical Perspectives in Sagebrush Steppe Ecosystems

SAGEBRUSH SYSTEMATICS AND DISTRIBUTION

E. Durant McArthur

INTRODUCTION

In this paper on sagebrush systematics and distribution, it is appropriate to begin by defining a few terms. Sagebrush, under my definition, are woody North American *Artemisia* of the subgenus *Tridentatae*. *Tridentatae* are one of four subgenera in *Artemisia*. *Tridentatae* or true sagebrush are separated from other *Artemisia* of the subgenera *Artemisia*, *Dracunculus*, and *Seriphidum* (e.g., wormwood, wormseed, sage, tarragon, etc.) by their completely woody nature, exclusive North American distribution, distinctive chemistry and molecular genetics, and their fertile, homogamous, perfect disc flowers (McArthur 1979, McArthur and Sanderson 1999a). There are 11 sagebrush species that, together with their subspecific entities, account for about 20 taxa. *Artemisia* as a whole includes more than 200 species.

We define systematics following Judd et al. (1999): systematics is the science of organismal diversity which entails the discovery, description, and interpretation of biological diversity as well as the synthesis of information in the form of predictive classification systems. According to Judd et al. (1999) the aim of systematics is to discover the branches of the tree of life, to document the changes that have occurred during the evolution of these branches, and to describe taxa (usually species) at the tips of these branches.

Distribution, of course, is the natural geographic range of organisms. For sagebrush taxa, there is a distribution of the whole group and subset distributions of taxa that constitute sagebrush which may be sympatric (occurring in the same area), parapatric (occurring in separate but adjoining areas), and allopatric (occurring in different areas).

SAGEBRUSH SYSTEMATICS

Artemisia is a distinguished name, an etymological descendant of an early Mother Nature. Artemis was the ancient Greek goddess of wild animals, the hunt, and vegetation, and of chastity and childbirth (McArthur 1979). *Tridentatae* and *tridentata* both refer to the characteristic three lobes of many sagebrush taxa.

Subgenus *Tridentatae* of *Artemisia* is a group of plants centered on the landscape-dominant *A. tridentata* complex. There have been several systematic treatments of the group (see Kornkven et al. 1998 and McArthur et al. 1998*a* for recent reviews). My colleagues and I (McArthur et al. 1998*a*, McArthur and Sanderson 1999a) recognize 11 species and 14 subspecies (Table 1). *Artemisia* is centered, in distribution and diversity, on the great Eurasian landmass. There is compelling distributional, chemical, and genetic evidence that North American *Tridentatae* are derived from Eurasian stock and that they differentiated and expanded during Pliocene and Pleistocene with the changing climates and habitats of those epochs (summarized in McArthur et al. 1998*a*,*b*; McArthur and Sanderson 1999a).

Differentiation and evolution within *Tridentatae* has been facilitated by polyploidy and hybridization. All the major species (big sagebrush [*A. tridentata*], silver sagebrush [*A. cana*], low sagebrush [*A. arbuscula*], and black sagebrush [*A. nova*]), as well as several less common or more geographically restricted ones (Bigelow sagebrush [*A. bigelovii*] and Rothrock sagebrush [*A. rothrockii*]), include both diploid and polyploid populations (Table 2). Based on habitat occupation, we have hypothesized that polyploidy is adaptive, i.e., polyploid populations are usually found in drier habitats than are related diploids (Sanderson et al. 1989, McArthur and Sanderson 1999a). Polyploids are smaller with slower growth rates that make them better adapted to drier conditions (Sanderson et al. 1989).

Some species have poorer support for taxonomic placement in *Tridentatae* than others. Bigelow sagebrush has floral anomalies, and pygmy sagebrush (*A. pygmaea*) has morphological anomalies; but both have karyotypic and molecular genetic characteristics of *Tridentatae*. There is evidence that sand sage (*A. filifolia*), ordinarily placed in subgenus *Dracunculus*, has some affinities with subgenus *Tridentatae* based on chloroplast DNA, plant chemistry, and chromosomal karyotype (Kornkven et al. 1998, McArthur and Sanderson 1999a). An anomalous plant, *A. palmeri*, is wholly herbaceous but has the floral formula of *Tridentatae*; however, I follow Rydberg (1916) and exclude it from *Tridentatae*.

Hybridization is common in this group and has apparently been a mechanism providing new genetic combinations to facilitate occupation of changing habitats during the evolutionary history of *Tridentatae* (Ward 1953, McArthur et al. 1988, McArthur and Sanderson

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1999a). In a series of studies (reviewed by Graham et al. 1999 and McArthur and Sanderson 1999b), my colleagues and I have examined a narrow hybrid zone between basin (*A. tridentata* ssp. *tridentata*) and mountain (*A. t.* ssp. *vaseyana*) big sagebrush. We have studied adaptation, growth, gene flow, chemistry, physiology, soils, mineral distribution and uptake, and plant and animal communities across the zone and in reciprocally transplanted gardens. We concluded that hybrids are adapted to these zones and that points of contact between differentiated taxa (hybrid zones) could have been the source for differentiation of new genetic combinations. These combinations were able to exploit new habitats associated with changing climates of the Pliocene and Pleistocene Epochs, continuing until the present. Several extant *Tridentatae* species, subspecies, and populations, described and undescribed, are of hybrid origin, e.g., Lahonton low sagebrush (*A. arbuscula* ssp. *longicaulis*), spicate or snowbank big sagebrush (*A. tridentata* ssp. *spiciformis*), and xeric big sagebrush (*A. tridentata* ssp. *xericensis*) (Winward and McArthur 1995, McArthur and Sanderson 1999b). Artificial hybridization may be useful for management purposes in selecting and combining traits in sagebrush for palatability, nutritive quality, and fire tolerance (McArthur et al. 1988, McArthur et al. 1998*a*).

SAGEBRUSH DISTRIBUTION

Artemisia in general is widely distributed throughout the northern hemisphere with disjunct distribution to some, mainly mountainous, southern hemisphere locations (Good 1974). However, the subgenus *Tridentatae* is wholly western North American (Fig. 1). West of the 100° west longitude at mid-latitudes, sagebrush is a dominant, widely distributed plant (Fig. 1, Table 1). Figure 1 illustrates aspects of the group's distribution, including the wide distribution of the central species, big sagebrush. Big sagebrush, with its subspecies, extends over most of the geographic range covered by the subgenus as a whole. The nature of areas dominated by sagebrush is also illustrated in the figure, using the state of Utah as an example. Large areas are dominated by sagebrush, but some of its species are less significant components of other communities, e.g., mountain brush and pinyon-juniper. However, some taxa, e.g., pygmy sagebrush, Bigelow sagebrush, Alkali sagebrush (*A. longiloba*), and stiff sagebrush (*A. rigida*), grow in specific, limited habitats.

Sagebrush taxa grow at elevations and precipitation levels above the salt desert shrub communities, i.e., precipitation above 18-20 cm per year. For the common big sagebrush subspecies in the Intermountain area, the annual precipitation levels are: about 32-36 cm for basin big sagebrush (however, basin big sagebrush often grows in areas that benefit from seasonally high water tables at different precipation levels), about 20-30 cm for Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*), and generally above 30 cm for mountain big sagebrush (Monsen and McArthur 1984, Goodrich et al. 1999).

General distribution and site conditions for each species and subspecies are presented in Table 1. Distribution of sagebrush species and subspecies is usually associated with specific soil properties and soil parental material as well as climatic differences (Passey et al. 1982, Wang et al. 1998, Wang et al. 1999). Seed recruitment conditions are generally tied to local climatic conditions, i.e., seeds germinate and establish better in habitats climatically like those that produced them (Meyer and Monsen 1992).

There is a high incidence of parapatric and sympatric distribution within *Tridentatae*. Many taxa, however, have allopatric distributions with regard to one another (Table 1). This type of distribution pattern, together with wind pollination, facilitates hybridization within the group. However, despite hybridization and the occurrence of hybrid zones, most populations and individuals are clearly assignable to parental taxa (Beetle 1970).

Unfortunately, sagebrush ecosystems have been badly disturbed (intensive grazing, introduction of cheatgrass, etc.) beyond historic natural disturbance cycles, as witnessed by other contributions in this symposium and by previous works, e.g., Passey et al. (1982) and contributions in Monsen and Kitchen (1994). I believe integrity and maintenance of sagebrush communities is important for healthy, naturally functioning ecosystems on a continental scale, as many other components of sagebrush ecosystems are dependent on this keystone species.

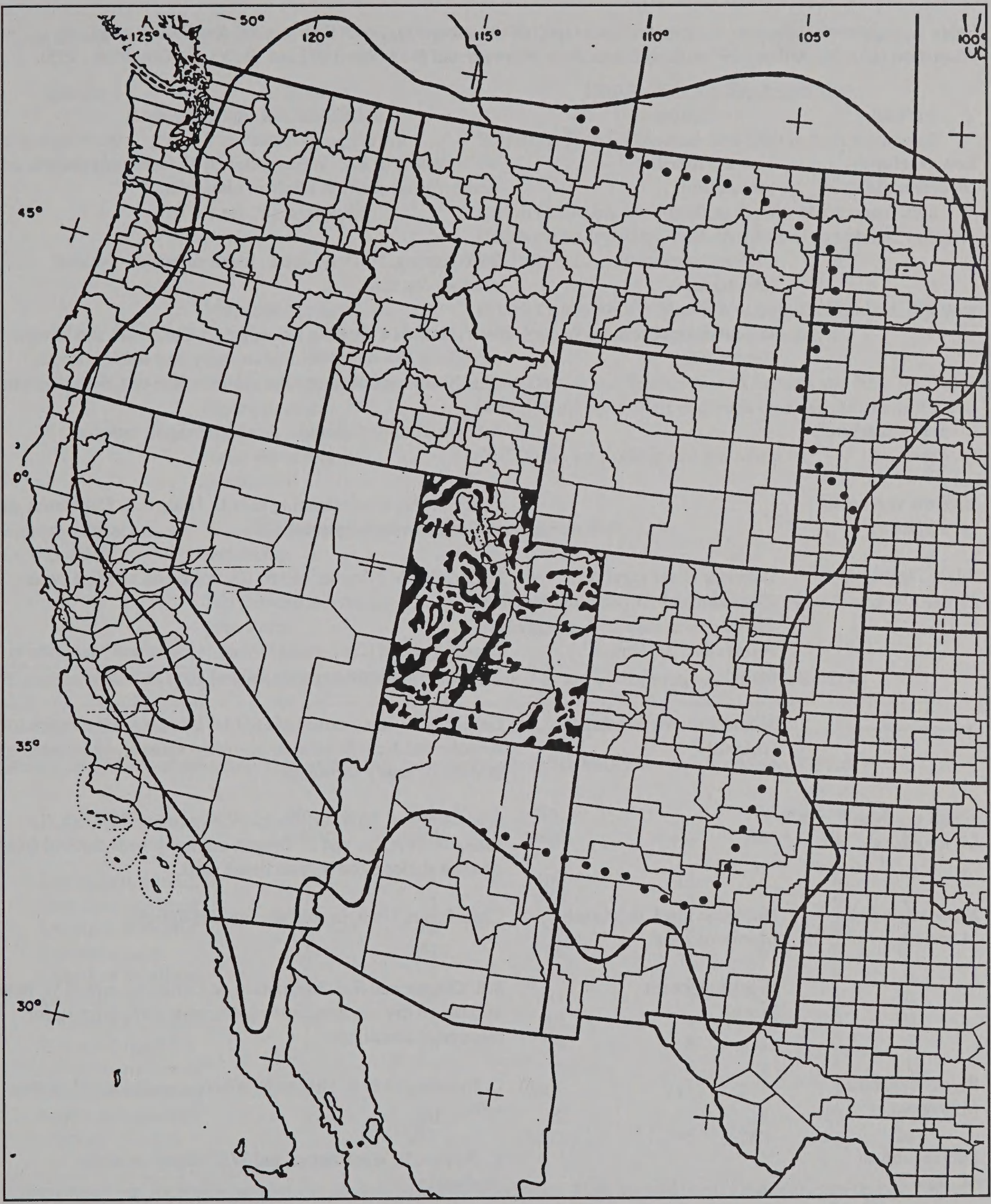

Figure 1. Distribution of sagebrush (subgenus *Tridentatae* of *Artemisia*). Solid line is the extent of distribution. Dotted lines delineate areas where *A. tridentata* doesn't grow; the northeast extension reflects the distribution of *A. cana*, and the southeast extension reflects the distribution of *A. bigelovii*. Solid black areas are zones of sagebrush dominance in Utah (from McArthur 1979).

Table 1. Sagebrush (subgenus *Tridentatae*) taxa (species and subspecies) with their general distributions and site adaptation (after McArthur 1994 with additions from Winward and McArthur 1995 and Welsh and Goodrich 1995).

Species	Subspecies	Distribution and Site Adaptation
Low sagebrush (*A. arbuscula*)	Low sagebrush (*arbuscula*)	W. Wyoming to S.C. Washingon and N. California on dry, sterile, rocky, shallow, alkaline, clay soils.
	Cleftleaf sagebrush (*thermopola*)	W. Wyoming, N. Utah, and E. Idaho on spring-flooded, summer-dry soils.
	Lahontan sagebrush (*longicaulis*)	N.W. Nevada extending into adjacent California and Oregon on soils of low water-holding capacity and shallow depth, usually around and above the old shoreline of Lake Lahontan.
Coaltown sagebrush (*A. argillosa*)		Jackson County, Colorado, on alkaline spoil material.
Bigelow sagebrush (*A. bigelovii*)		Four Corners area extending to N.E. Utah, S.E. California, and W. Texas on rocky, sandy soils.
Silver sagebrush (*A. cana*)	Bolander silver sagebrush (*bolanderi*)	E. Oregon, W. Nevada, and N. California on alkaline basins.
	Plains silver sagebrush (*cana*)	Generally E. of Continental Divide, Alberta and Manitoba to Colorado on loamy to sandy soils of river bottoms.
	Mountain silver sagebrush (*viscidula*)	Generally W. of Continental Divide, Montana and Oregon to Arizona and New Mexico in mountain areas along streams and in areas of heavy snowpack.
Alkali sagebrush (*A. longiloba*)		S.W. Montana, N.W. Colorado, W. Wyoming, N. Utah, S. Idaho, N. Nevada, and E. Oregon on heavy soils derived from alkaline shales or on lighter, limey soils.
Black sagebrush (*A. nova*)	Duchesne black sagebrush (*duchesnicola*)[a]	Uinta Basin, Utah, in reddish clay soil uplands.
	Black sagebrush (*nova*)	S.E. Oregon and S.C. Montana to S. California and N.W. New Mexico on dry, shallow, stony soils, with some affinity for calcareous conditions.
Pygmy sagebrush (*A. pygmaea*)		C. Nevada and N.E. Utah to N. Arizona on desert calcareous soils.
Stiff sagebrush (*A. rigida*)		E. Oregon, E. Washington, and W.C. Idaho on rocky scablands.
Rothrock sagebrush (*A. rothrockii*)		California and Nevada in deep soils along the forest margins of the Sierra Nevada and outliers.

(Continued)

Table 1 (cont.)

Species	Subspecies	Distribution and Site Adaptation
Big sagebrush (*A. tridentata*)	Snowbank big sagebrush (*spiciformis*)	Wyoming, Idaho, Colorado, and Utah in high mountains.
	Basin big sagebrush (*tridentata*)	British Columbia and Montana to New Mexico and Baja California in dry, deep, well-drained soils on plains, valleys, and foothills.
	Mountain big sagebrush (*vaseyana*)	British Columbia and Montana to Baja California in dry, deep, well-drained soils on foothills and mountains.
	Wyoming big sagebrush (*wyomingensis*)	North Dakota and Washington to Arizona and New Mexico in poor shallow soils often underlain by a caliche or silica layer.
	Xeric big sagebrush (*xericensis*)	W.C. Idaho on basaltic and granitic soils.
Threetip sagebrush (*A. tripartita*)	Wyoming threetip sagebrush (*rupicola*)	Wyoming on rocky hills.
	Tall threetip sagebrush (*tripartita*)	E. Washington and W. Montana to N. Nevada and N. Utah on moderate-to-deep well-drained soils

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[a] Described at the variety level by Welsh and Goodrich (1995) but analogous to the other subspecies listed in the table.

Table 2. Summary of subgenus *Tridentatae* chromosome counts (after McArthur and Sanderson 1999a).

Species	No. ssp.[a]	No. pops.	No. plants	No. pops.[b] at			
				2x	4x	6x	8x
Artemisia arbuscula[b]	2	51	139	25	18	8	0
Artemisia argillosa	1	1	4	0	1	0	0
Artemisia bigelovii[b]	1	12	46	4	7	0	1
Artemisia cana	3	43	96	13	6	0	24
Artemisia longiloba	1	3	8	2	1	0	0
Artemisia nova[b]	1	36	81	13	23	0	0
Artemisia pygmaea[b]	1	4	12	4	0	0	0
Artemisia rigida[b]	1	13	30	8	5	0	0
Artemisia rothrockii[b]	1	7	8	0	2	4	1
Artemisia tridentata[b]	5	427	1103	213	214	0	0
Artemisia tripartita[b]	1	20	46	14	6	0	0
Totals		617	1573	296	283	12	26

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[a] Includes only ssp. for which chromosome numbers have been determined. There are additional subspecific taxa that are cytologically unknown: *A. arbuscula* ssp. *thermopola*, *A. nova* var. *duchesnicola*, *A. tripartita* ssp. *rupicola*.

[b] Some populations have plants at more than one chromosome ploidy level. The ploidy (x) level reported here is that of the mode of the sampled population(s) or the lowest number when an equal number of plants were at different x levels.

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SYNECOLOGY AND DISTURBANCE REGIMES OF SAGEBRUSH STEPPE ECOSYSTEMS

Neil E. West

ABSTRACT

The pre-Columbian mixed-growth form, composition, and structure of sagebrush steppes was mostly due to the highly variable semiarid climate and long fire-free intervals. The weak stability of this relatively complex vegetation was easily upset by excessive livestock grazing, especially in drought periods. After a few decades of uncontrolled livestock grazing, it was easy for introduced winter annuals, especially cheatgrass, to dominate the understory and alter the fire regime to larger, more frequent fires that occur earlier in the year. Accelerated soil erosion has caused many sites to lose the potential for management back toward native perennial dominance by controlling only livestock and fire. Major investments will probably be necessary to lengthen the current fire-free interval, as well as reduce the size of fires and their occurrence during late spring and early summer on large areas of cheatgrass dominance. Livestock could be used in some circumstances to help reverse the damage they did before grazing became regulated. Opportunities to apply genetic engineering to native plants and new herbicides to cheatgrass should also be explored before even more noxious biennials gain a major foothold.

INTRODUCTION

Durant McArthur (this volume) appropriately began by giving us background in sagebrush taxonomy, species distributions, and autecology. I now perceive my role as one of reviewing the synecology of an ecosystem type called “sagebrush steppe.” This includes the disturbance regimes intrinsic to this ecosystem.

DEFINITIONS

I have restricted my coverage to the 45 million ha of sagebrush steppe (West 1983*a*) and alert you to the fact that not all areas currently or recently having vegetation with a woody *Artemisia* dominant are sagebrush steppe, particularly in the drier, less diverse, less productive, less resistant, less resilient sagebrush semi-desert to the south (West 1983*b*). I am purposely avoiding drawing on information from sagebrush semi-deserts in this paper.

ECOSYSTEM STRUCTURE

Climate - Sagebrush steppe occurs where there has been until recently, or still is, a sharing of dominance between shrub and herbaceous growth forms. The fundamental reason for this is that, on average, continental semiarid climates occur here. More important than the climatic means is the understanding that these climates have high coefficients of variation (~30%) in total annual precipitation, with rapid fluctuation between some more favorable years that promote the shallow, fibrous-rooted, herbaceous plants and droughty years that favor the more deeply rooted shrubs (Fig. 1). Herbaceous plants develop earlier in the growing season and thrive on spring rains, whereas shrubs lag in their phenological development because they can draw from deeply infiltrating moisture from snowmelt the previous fall and winter. While this leads to some compensation between species to produce a dampened yet higher level of production in shrub steppes than in semi-deserts, it also makes these systems much more difficult to understand and sustainably manage than either grassland or desert.

The fire-return interval in the Pre-Columbian condition probably varied between 25 years in wetter areas (Houston 1973) and 110 years on the central Snake River Plains (Whisenant 1990) (Fig. 2). Otherwise, the earliest observers would have called this the rabbitbrush steppe because the shorter-lived and root-sprouting *Chrysothamnus* spp. would have prevailed (Young 1983).

Soils - Soils give us some reflection of long-term climatic and vegetational influences. Most sagebrush steppe soils are Xerolls – that is, the most drought-affected Mollisols – if the surface layers haven't been eroded. Most soils of sagebrush semi-desert are Aridisols (West and Young 2000). Thus, where flora and fauna are highly altered, one can use soil profile characteristics to gauge the potential of sites for recovery through management or restoration.

Vegetation - The floristic diversity of the sagebrush steppe is moderate by regional standards. Daubenmire (1970) found an average of 20 vascular plant species in 1,000-m^2 plots on relict sites in central Washington. Tisdale et al. (1965) found from 13 to 24 vascular plant species in examples of three community types on an ungrazed site in southern Idaho. Mueggler (1982) found 24 to 41 vascular plant species in a set of 68 lightly

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grazed macroplots in the sagebrush steppe of western Montana.

The vertical and horizontal structure of the sagebrush steppe consists of shrub-dominated and herb-dominated phases (West 1983*a*). The shrubs usually vary in height from about 0.5 m for either young plants of the tall sagebrushes or mature low-statured species to more than 2 m for the tallest sagebrushes on the best sites. The fraction of ground surface covered by the various growth forms varies greatly depending on site and successional status.

Herbs on relict sagebrush steppe sites are usually perennial hemicryptophytes (Daubenmire 1975). The proportion of geophytes approaches 20%. Bork et al. (1998) claim that grasses are more often situated closer to the shrubs than the forbs. Annuals and microphytes are usually more abundant in the middle of the interspaces between shrubs.

The total phytomass standing crop of relictual stands varies between 2 and 12 t/ha, with about half of that occurring below ground. Only about 15% of the above-ground phytomass may be attributable to the current year's growth of shrubs. Above-ground net primary production varies from about 100 to 1,500 kg/ha/yr for relict areas (Passey et al. 1982).

Animals - Native vertebrate animals of the sagebrush steppe are a mixture of grassland and desert species. About 100 bird and 70 mammal species can be found in sagebrush habitats (Braun et al. 1976). Although the vertebrate community is most diverse when the pattern of plant communities is most structurally diverse (Parmenter and MacMahon 1983, Maser et al. 1984), the only tightly co-evolved and thus sagebrush obligate vertebrate species are the sage grouse, sage sparrow, Brewer's sparrow, sage thrasher, pygmy rabbit, sagebrush vole, sagebrush lizard, and pronghorn (Paige and Ritter 1999). While none of these is known to cause major negative feedbacks on the vegetation, jackrabbits can (Young 1994).

Over 1,000 species of insects have been found on example sites (West 1999), more than 76 species on sagebrush alone (Wiens et al. 1991). While some are known to alter the vegetation during occasional population explosions, e g., *Aroga* moth and cicadas (West 1999), grasshoppers and crickets (Yensen 1980) can do so more regularly. The functional importance of most invertebrates is yet to be discovered.

Microbes - We know very little about what microbes are present and how they influence ecosystem processes within the sagebrush steppe. Hopefully, these organisms and the work they do, mainly decomposition and nutrient cycling, will receive more attention in the future. Global environmental changes are likely to produce some unexpected interactions among plants, microorganisms, and soil degradation (West et al. 1994).

ECOSYSTEM DYNAMICS

We now need to turn to consideration of how the above components interact and how the ecosystem has changed. In order to interweave the historical with the ecological, I will follow the recent example of Rapport and Whitford (1999) in organizing this overview of how sagebrush steppes have responded to stress. I will also tie the changes to a recent model of retrogression in the sagebrush steppe (West 1999). Only the major states and pathways are considered here.

Pristine Conditions (State I)

Pristine ecosystems (State I in Fig. 3) no longer exist, nor are they likely to be recoverable. The reasons for this view are:

1. Humans (indigenous peoples) are no longer hunting, gathering, and burning the areas. The previous fire regimes are no longer in place; and, as the vegetation has changed in response to fires, the hydrologic and nutrient cycles have been altered, as has the habitat for numerous animals and microbes.
2. The present climate is warmer and drier than the cooler, wetter Little Ice Age climate which prevailed from about 1500 to 1890. Thus, only heat- and drought-tolerant species may now thrive under global warming.
3. Atmospheric CO_2 has increased about 20% during the past century, altering the competitive balances in this vegetation as well as changing the nutritional qualities of the phytomass and litter (Polley 1997).
4. About 15% of the flora is new to the region. Since the close of the Pleistocene, extinctions have been minor.

Since we can reverse none of these influences, at least in the short term, we should learn to live with what remains and manage it toward the mix of desired plant communities we choose for each landscape (Paige and Ritter 1999).

Relictual Conditions (State II)

There are some remnants of the present landscapes that have escaped direct human influences. These relicts exist because they have no surface water, are surrounded by difficult topography, or are protected in special-use areas, e.g., Research Natural Areas. I place these in State II (Fig. 3). Passey et al. (1982) describe many examples. These relicts are not completely reliable as reference conditions because they are incomplete ecosystems. They lack indigenous humans as well as normal kinds and numbers of native animals and have usually experienced lengthened fire frequencies because of their isolation. Relicts are further influenced by air pollutants, climatic change, and invasion by exotics (Passey et al. 1982).

Most of the existing late seral sagebrush steppe with good perennial understory (State II in Fig. 3) has had light livestock use, especially earlier in the century when sheep were very abundant. Even light livestock use (T_1) puts inordinate pressure on a few highly palatable species ("ice cream plants"), partially explaining the lack of a return arrow from State II to State I. I estimate that less than 1% of the region remains in State II (Fig. 4). These shrub steppes with smaller, more scattered shrubs and almost complete perennial herbaceous understories are less susceptible to large-scale fires and subsequent invasion by cheatgrass (Peters and Bunting 1984).

Stagnant Sagebrush (State III)

Because livestock that graze native sagebrush steppe tend to avoid the unpalatable species (usually woody species), shrubs are freed from competition and achieve dominance quickly (10-15 years). With the removal of fine connecting fuels, the chance of fire is also reduced in State III (Fig. 3). About 25% of this ecosystem type is estimated to exist in this state (Fig. 4). In some places, feral horses, protected by law on most public lands, have created and maintain sagebrush stands with little remaining herbaceous perennial understory. Most of these stands can remain stagnant for decades (Rice and Westoby 1978, Sneva et al. 1984, Winward 1991). The dense, competitive stands of excess sagebrush prevent perennial herbaceous species from recovering when grazing is either reduced (T_3) or excluded over very long intervals (Bork et al. 1998).

Herb-dominated Stands (State IV)

Brush-choked or stagnant stands of sagebrush (State III) were usually chosen by both livestock and wildlife managers in the past for manipulation to diversify vegetation structure. Such treatments locally enhance a stand by concentrating livestock use and reducing pressure elsewhere, while simultaneously creating an advantage for some wildlife species through vegetation modifications via grazing systems, prescribed burning, brush-beating, or chaining (T_3). For example, grazing sheep only in the fall – because they consume more sagebrush then but cannot heavily impact the herbs – can help achieve a conversion from State III to State IV and even increase floristic diversity compared to adjacent exclosures ungrazed for decades (Bork et al. 1998). Prescribed burning (Harniss and Murray 1973) can also be applied to stands with sufficient remnant populations of perennial native herbs to quickly recover following brush kill. A rest-rotation grazing system or winter-only use (Mosley 1996) will often allow a slow return (T_6) to State II from State IV.

Reduction of brush also enhances water yields (Sturges 1977), and some seeps, springs, and streams reappear. When phenoxy herbicides are used alone (Evans et al. 1979) (T_4) or in conjunction with fire, the community becomes dominated by native grasses (State IV, Fig. 3) because phenoxy herbicides negatively impact all broad-leafed species. This conversion slowly returns (T_6) to State II only with conservative grazing. About 5% of the remaining sagebrush steppe is now estimated to be in State IV. This is a short-lived state, especially under heavy grazing (T_5). Mueggler (1982) found enhanced alpha diversity in moderately grazed sagebrush steppe communities in western Montana following prescribed fire, 2,4-D, and brush-beating treatments. Summer fires can damage some grass species (Young 1983) but encourage the resprouting rabbitbrushes (*Chrysothamnus* spp.) and horsebrushes (*Tetradymia* spp.) (Anderson et al. 1996).

The perceived will of a majority of Americans now is to identify remaining areas occupied by States II and III, especially those on public lands, and protect them from development. In other words, I agree with Paige and Ritter (1999) that no net loss of sagebrush should be a regional objective to prevent further declines in biodiversity (West 1999). Some advocate all such areas have livestock removed (Kerr 1994), whereas others (Bock et al. 1993) propose that 25% have livestock excluded. Rose et al. (personal communication) have, however, recently demonstrated that lightly grazed sagebrush steppe has higher species richness than adjacent exclosures dating to 1937. Others propose restoration efforts to bring further-degraded systems back to States I or II. Whether that is possible and economical is discussed in the remainder of this volume.

Regardless of one's view of the matter, State II and III areas will serve as a major "parts catalog" for restoration efforts. The Gap Analysis Program (GAP) of the U.S. Fish and Wildlife Service (Scott et al. 1993) and the various natural heritage programs initiated by the Nature Conservancy are well under way to identify such areas.

I expect to see physical modifications for enhancing production of food and fiber (formerly called range "improvements") to be more spatially limited than in the past. Such actions on public lands or with public monies on private land require environmental assessments or impact statements and, thus, public scrutiny and debate. The remaining sagebrush-dominated public lands will probably be consciously protected to provide the later seral condition patches necessary to hold a broader spectrum of all species and meet the special requirements for some featured and obligate species (Paige and Ritter 1999).

Rangeland managers in the past strove to reduce the land's limitations for producing livestock. These limitations were mainly topography, forage availability, and water. For example, trails were constructed into areas where topographic breaks limited previous livestock access. Natural water was supplemented by developing springs, building stock tanks and small dams, drilling wells, and piping and hauling water. Fences were constructed and salt distributed to control livestock movement and institute grazing management systems (e.g., rest-rotation grazing). All these "improvements" were designed to distribute livestock utilization more uniformly across the land, gain greater efficiency of food and fiber production, and divert livestock from the especially sensitive riparian areas (Elmore and Kauffman 1994, Laycock 1995). The net result has been progressively more widespread yet intensive use of a landscape that has become at least partially tamed from the wild. These assumptions need to be reexamined in the light of biodiversity concerns. Let us continue our consideration of these relationships in the mostly highly altered sagebrush steppe areas.

If accelerated soil erosion does not ensue and the fundamental potential of the site does not change, then State III can be maintained or managed toward States II or IV. However, as herbaceous plants, litter, and microphytes in the interspaces between perennials are reduced, soil aggregate stability declines, infiltration of precipitation diminishes, overland flow increases, and soil erosion frequently increases (Blackburn et al. 1992). When a probable threshold is exceeded, the site can irreversibly change to one of lesser potential. This explains the dashed line and downward arrows below States III and IV as permanent transitions, where the syndrome of desertification is most evident.

All the previously discussed states shown above the dashed line of Fig. 3 can be dealt with via management approaches using "soft" energy. Once this threshold is exceeded, however, subsequent management requires expensive, risky, "hard" energy solutions. Unfortunately, it is often easier to get political attention after major damage has been done rather than getting budgets and personnel to plan, monitor, and tweak the healthier, more natural systems at opportune times.

Desertified Sagebrush Steppe (State V)

The desertified sites are usually initially dominated by taller, thickened brush and have largely introduced annuals in their understory. The major adventive from 1870 onward has been cheatgrass (*Bromus tectorum*) (Billings 1990, Knapp 1996). I estimate that State V comprises about 25% of the current sagebrush steppe region (Fig. 4). Removal of livestock usually only hastens further degradation from State V because livestock remove part of the herbaceous fuel load and thus reduce the chance of fire destroying the sagebrush and the spots of enriched soil it protects (Charley and West 1975). Cheatgrass fundamentally changes the fire regime (Fig. 2), and most sagebrushes, not being root sprouters, only return slowly, if ever. Livestock can be used in the spring to reduce cheatgrass (Mosley 1996); however, grazing at that time also impacts any remaining native herbs. Where there are warm season (C_4) grasses and forbs, heavy livestock grazing in the spring with deferment in summer can be used to favor the recovery of those components (R. Budd, personal communication, 1999).

Introduced Bunch Grasslands (State VI)

If insufficient amounts of native grass remain in the sagebrush steppe to allow a reasonably short return to other desired plant communities, the usual response by land management agencies has been to destroy the sagebrush and replace it mechanically (T_7) with introduced wheatgrass and ryegrass, especially crested wheatgrass (Asay 1987). This has been done because the seed of introduced perennial grasses is more readily available and less expensive and their seedlings are much more easily established than the native grasses. They also grow quickly to provide more forage with a higher nutritional plane. The introduced perennial grass stands are also much more tolerant of subsequent heavy livestock use and last for many decades (Johnson 1986). There are some long-range concerns, however (Lesica and DeLuca 1996), because the introduced perennial grasses suppress the return of natives and, thus, richer plant species assemblages. Some large treatment areas are essentially monocultures of Eurasian perennial grasses (State VI, Fig. 3). I estimate about 5% of the original sagebrush steppe has already been transformed to State VI (Fig. 4).

Wildlife biologists have noted declines in the numbers of birds (Olson 1974; Reynolds and Trost 1979, 1981), small mammals (Reynolds and Trost 1979), and large reptiles (Reynolds 1979) on such seedings of introduced grasses in the sagebrush steppe area. It should be noted, however, that such studies present a worst-case scenario because samples came from the center of large treatments. Provision for increased diversity near edges (Thomas et al. 1979) is not usually mentioned in such studies. Present-day, more sensitized planners would provide for optimum edge effect and patchiness (McEwen and DeWeese 1987, Paige and Ritter 1999).

When society made the investment in repairing severely damaged sagebrush steppe, e.g., creating perennial grass-dominated pastures of species palatable to livestock (T_7) with much greater productivity, this compensated for livestock reductions and other management restrictions on lands where States II, III, and IV (Fig. 3) predominated. Because introduced grass pastures can take much heavier utilization in the spring than the native shrub steppe, livestock can be grazed on native sagebrush steppe in fall or winter with less impact, especially on the native herbaceous perennials.

Shrub-Reinvaded Introduced Grasslands (State VII)

Introduced perennial grass plantings in the sagebrush steppe region, especially if grazed by livestock, will eventually experience shrub reinvasion (T_8 to State VII, Fig. 3), largely in response to intensity and timing of livestock grazing. I estimate (Fig. 4) that about 5% of the sagebrush steppe region is currently represented by shrub-reinvaded introduced wheatgrass/ryegrass pastures (State VII).

Shrubs reinvading State VII are not being eliminated by herbicides, as was once attempted. All herbicide use in such circumstances on public lands has been suspended by judicial decree in the Pacific Northwest. Prescribed burning of the coarser, introduced grasses is difficult and leaves patches where the shrubs prevail. Therefore, there are opportunities to enhance edge effects in large areas that were formerly homogenized. As in the untilled native areas, patchy burning could enhance wildlife habitat across landscapes by providing a mix of successional stages over a landscape, providing both cover and forage for either featured species or total species richness (Maser et al. 1984). For example, some success has been attained in creating alternate leks for sage grouse following disturbance (Eng et al. 1979). Some crested wheatgrass pastures on U.S. Forest Service lands in northeastern California have recently been plowed and planted with native herbs in an attempt to enhance biodiversity. Aggressive annuals such as yellow starthistle were the dominant result (J. Young, USDA ARS, personal communication).

Annual Grasslands (State VIII)

Despite greatly increased attention to fire prevention and control, much of the depauperate sagebrush steppe (State V) has been burned (T_{10}) at least once during the past three decades and is now almost completely replaced by introduced annuals, mainly grasses such as cheatgrass and medusahead (State VIII, Fig. 3). The Bureau of Land Management (M. Pellant, Bureau of Land Management, personal communication) estimates that about 3 million acres of public lands in Idaho, Utah, Oregon, and Nevada are now dominated by cheatgrass and medusahead. I estimate that about 25% of the total sagebrush steppe has made these transitions (T_{10}, T_{11}).

Because of their short stature, restricted nutritional characteristics (short period of above-ground greenness), and greater susceptibility to recurring fires and drought than sagebrush steppe, such areas are undesirable from all viewpoints (Knick and Rotenberry 1997). Without nutritional supplementation, livestock can graze State VIII only during the short, early-spring growing season. Winter use is possible only in the lower-elevation areas near the Columbia River (Mosley 1996). Only the most generalist animals, such as the introduced chukars, horned larks, grasshoppers, and deer mice, seem to thrive on the annual grasslands (Maser et al. 1984). When such areas burn in early summer, soils are bared to wind and water erosion during the convectional storms of summer. The consequent needs for revegetation after fire are increasing while the budgets of federal land management agencies decline and pressure increases from environmentalists who are against proactive management.

Land dominated by annuals may provide fair watershed protection during years without fire and actually appear to be more productive of total plant biomass than the original sagebrush-native perennial grass and forb combination (Rickard and Vaughn 1988). This is likely, however, to be only a temporary situation based on the priming effect of decomposing litter (Lesica and DeLuca 1996) and the mineralization of nutrients from the enormous below-ground necromass of the original system. The formerly strong link of net primary production with precipitation becomes decoupled (Whitford 1995). The shrub-centered islands of fertility (Charley and West 1975) are now diluted in a horizontal direction by the interactions of fire, soil erosion, and tillage. When these reserves of nutrients and soil organic matter are finally respired away, the annual grasslands are likely to become much less productive. Similar transitions happened in the Middle East several millennia ago (Zohary 1973). Many other more noxious weeds from that region could find their way here, and we could witness a downward spiral of further degradation (T_{12}).

REPAIRING THE DAMAGE

Rather than allowing the annual grasslands derived from former sagebrush steppe (State VIII, Fig. 3) to remain and the land to degrade further, some land managers are attempting to intervene. A joint program among the USDA Forest Service, Bureau of Land Management, Agricultural Research Service, and University of Idaho has been under way this past decade to reduce these threats (Pellant 1990). The most notable component of this effort is the greenstripping program, which is particularly evident in southern Idaho. The basic approach is to begin breaking up the now vast stretches of cheatgrass and other annual dominance that have developed as fires have become earlier, larger, and more frequent (Fig. 2). Land managers are attempting to break the cheatgrass-dominated areas into smaller, burnable units, especially in proximity to cities and towns. The approaches used thus far include planting strips of vegetation that stay green (and thus wetter and less burnable) longer than cheatgrass.

Although the introduced wheatgrasses, ryegrasses, and forage kochia (*Kochia prostrata*) do stay green longer and burn less readily because of coarser above-ground structure, they are not native and thus are rejected as replacements by some interest groups. Because the genetic biodiversity of the native plants is so primitively understood, the best that can be done is to gather such seed locally and plant it on comparable sites. Such seed sources are undependable, however. Thus, a root-sprouting big sagebrush is seen as a potentially better keystone species to put back in this area. A few sagebrushes may actually help sustain perennial grasses by harboring the predators on black grass bugs (*Labops* spp.) (Haws 1987). Furthermore, total plant community production can be enhanced (Harniss and Murray 1973) because sagebrushes help trap blowing snow (Sturges 1977) and scattered sagebrushes moderate temperatures (Pierson and Wight 1991), benefit the reestablishment of native herbs, and protect them from excessive utilization (Winward 1991). Sagebrushes also harbor mycorrhizal fungi (Wicklow-Howard 1989), which helps them extract nutrients from deep in the soil and recycle them to the surface through litter production (Mack 1977, West 1991).

Whether or not we can accomplish restoration of sagebrush steppe (T_{13}, between States V and III in Fig. 3) is highly questionable. Even where funding is less limiting and topsoil is replaced on coal strip mines, early results are only partially encouraging (Hatton and West 1987). We must learn much more about how sagebrush steppe ecosystems are structured and how they function, and we must have access to vast budgets and more trained personnel before such efforts are routinely successful. It is cheaper and more feasible to foster good stewardship of land having late seral vegetation (manage while in States I, II, III, or IV of Fig. 3) rather than rely on restoration efforts after degradation has taken place (States V, VI, VII, and VIII of Fig. 3).

The future of the sagebrush steppe region is the concern of this volume. Can the damage of the past be reversed or mitigated? Is restoration or rehabilitation possible and affordable? Remember that we have lost some pieces, gained new ones, and have a new and further changing environment. New invaders, increased temperatures, atmospheric CO_2, and UV_B pose additional problems.

While we must acknowledge that unrestricted livestock grazing, especially during droughts, was the fundamental cause of degradation of most sagebrush steppe, it doesn't automatically follow that reduction or even entire removal of livestock will reverse the changes for highly altered sagebrush steppe (below the dashed line in Fig. 3). Most of this land area has had threshold-exceeding changes. Soils, their nutrient pools and water handling capabilities, seed reserves, and thus their vegetation-producing potential have been fundamentally lowered. Even removing livestock during droughts will not suffice in attaining recovery. In fact, removal of livestock during wet years may increase the risk of wildfires, further damaging on-site features, as well as those at some distance, through wind erosion (dust storms). If livestock are totally removed, I predict we will have to eventually pay for them to return. The point is to constructively use them as tools within a holistically conceived recovery plan.

We must break the positive feedbacks, which allow further damage to the sagebrush steppe. The major linkage is between cheatgrass and larger, earlier, and more frequent fires (Fig. 2). I suggest further expansion of greenstripping with further use of the herbicide OUST® to reduce cheatgrass competition and allow better shrub establishment. A resprouting sagebrush would be desirable. If not that, rabbitbrushes are better than cheatgrass. Unpalatable strains of bluebunch wheatgrass (e.g., Whitmar) could be replanted to prevent overuse by livestock in the future. Let's enlist the genetic engineers to build us some perennial plants that better capture and conserve the resources that are truly irreplaceable – the soils. With the soils in place, future generations will have more options as new science and technology become available.

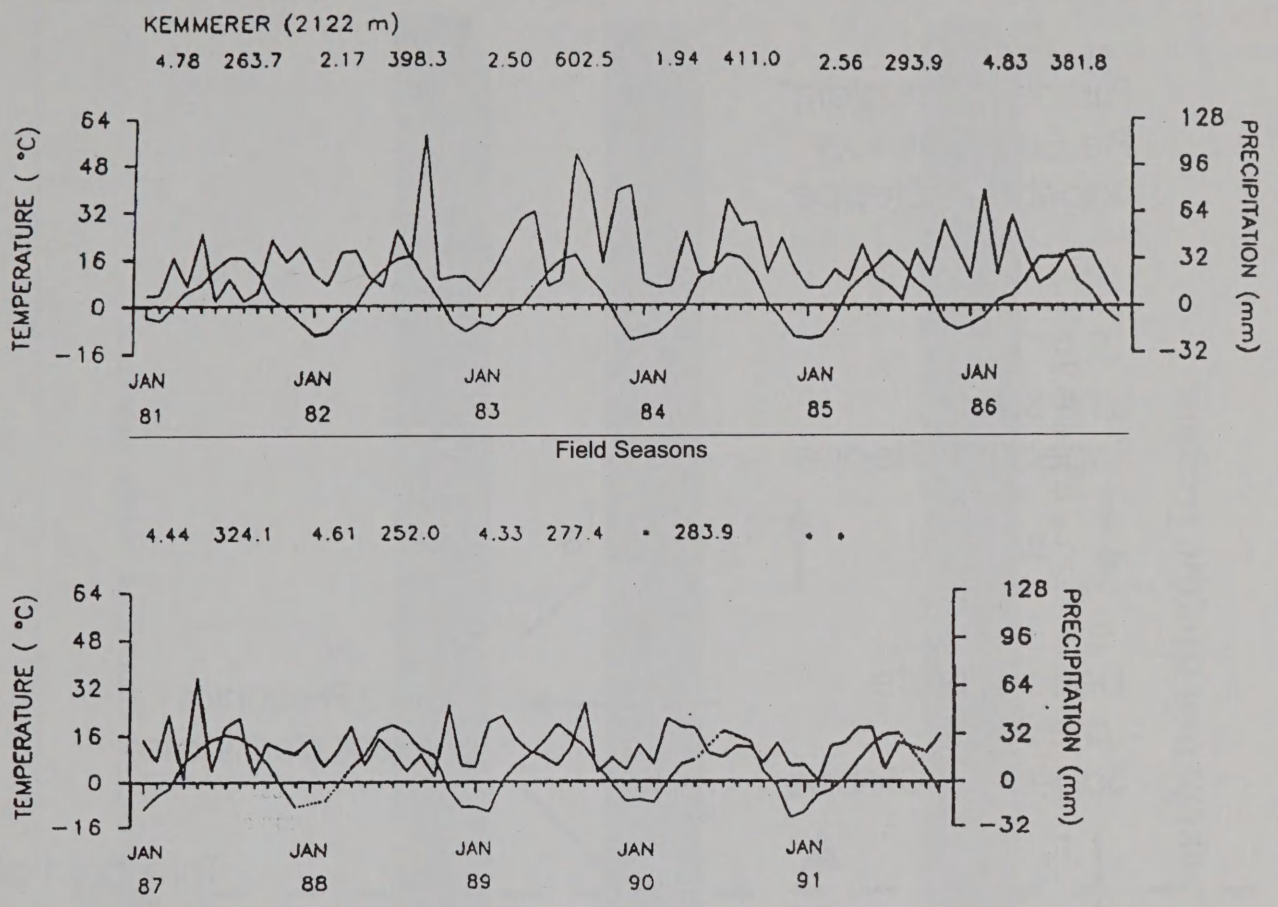

Figure 1. Patterns of mean monthly precipitation and temperature at Kemmerer, Wyoming, 1981 to 1991. Mean annual temperature (°C) and total annual precipitation (mm) are the numbers entered above the trend lines for each year.

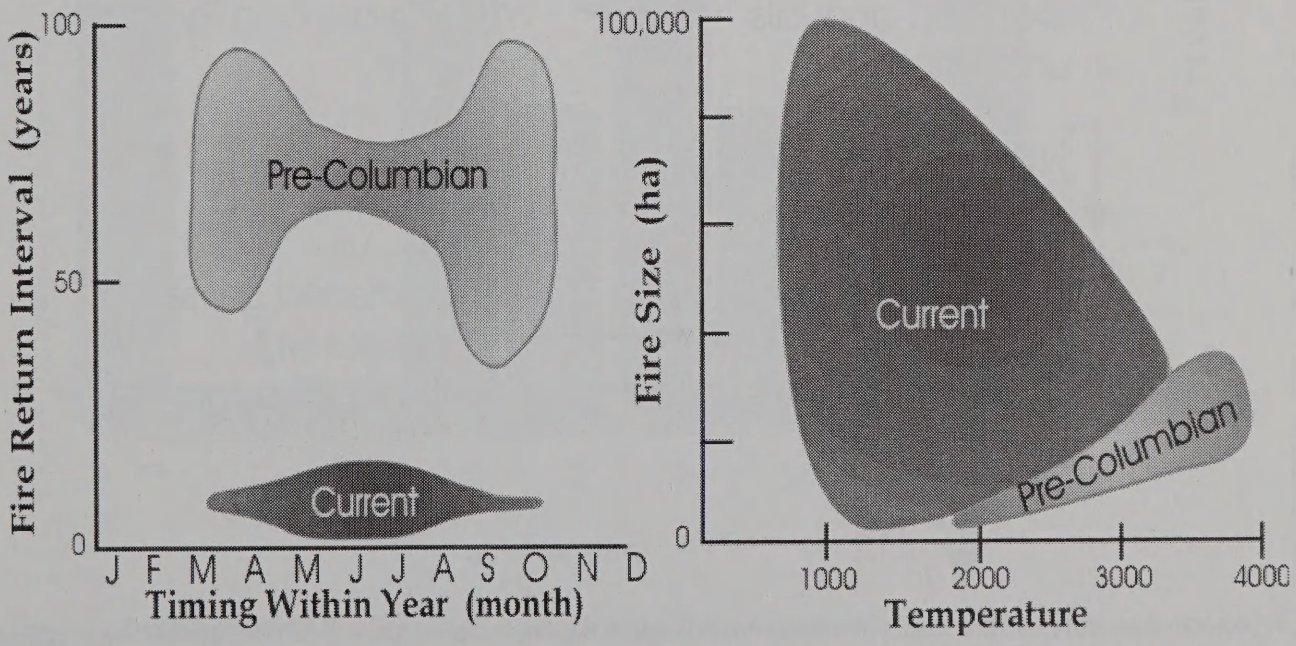

Figure 2. Estimated differences in the fire regime of sagebrush steppe in Pre-Columbian and current times.

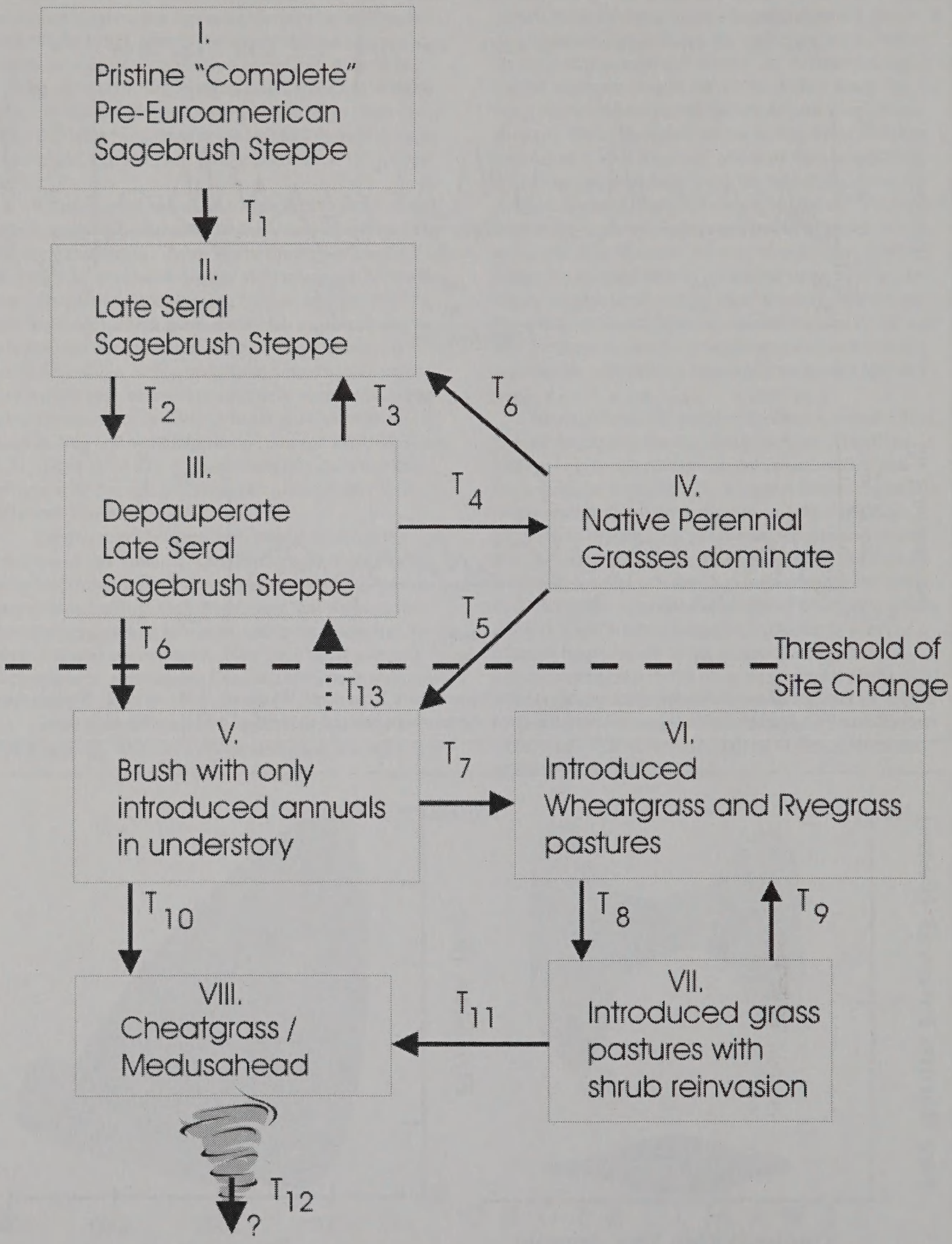
I.
Pristine "Complete"
Pre-Euroamerican
Sagebrush Steppe
T1
II.
Late Seral
Sagebrush Steppe
T2
T3
T6
III.
Depauperate
Late Seral
Sagebrush Steppe
T4
IV.
Native Perennial
Grasses dominate
T5
T6
Threshold of
Site Change
T13
V.
Brush with only
introduced annuals
in understory
T7
VI.
Introduced
Wheatgrass and Ryegrass
pastures
T10
T8
T9
VIII.
Cheatgrass /
Medusahead
T11
VII.
Introduced grass
pastures with
shrub reinvasion
T12
?

Figure 3. State and transition model of successional change in sagebrush steppe (from West 1999, permission to reprint from CRC Press).

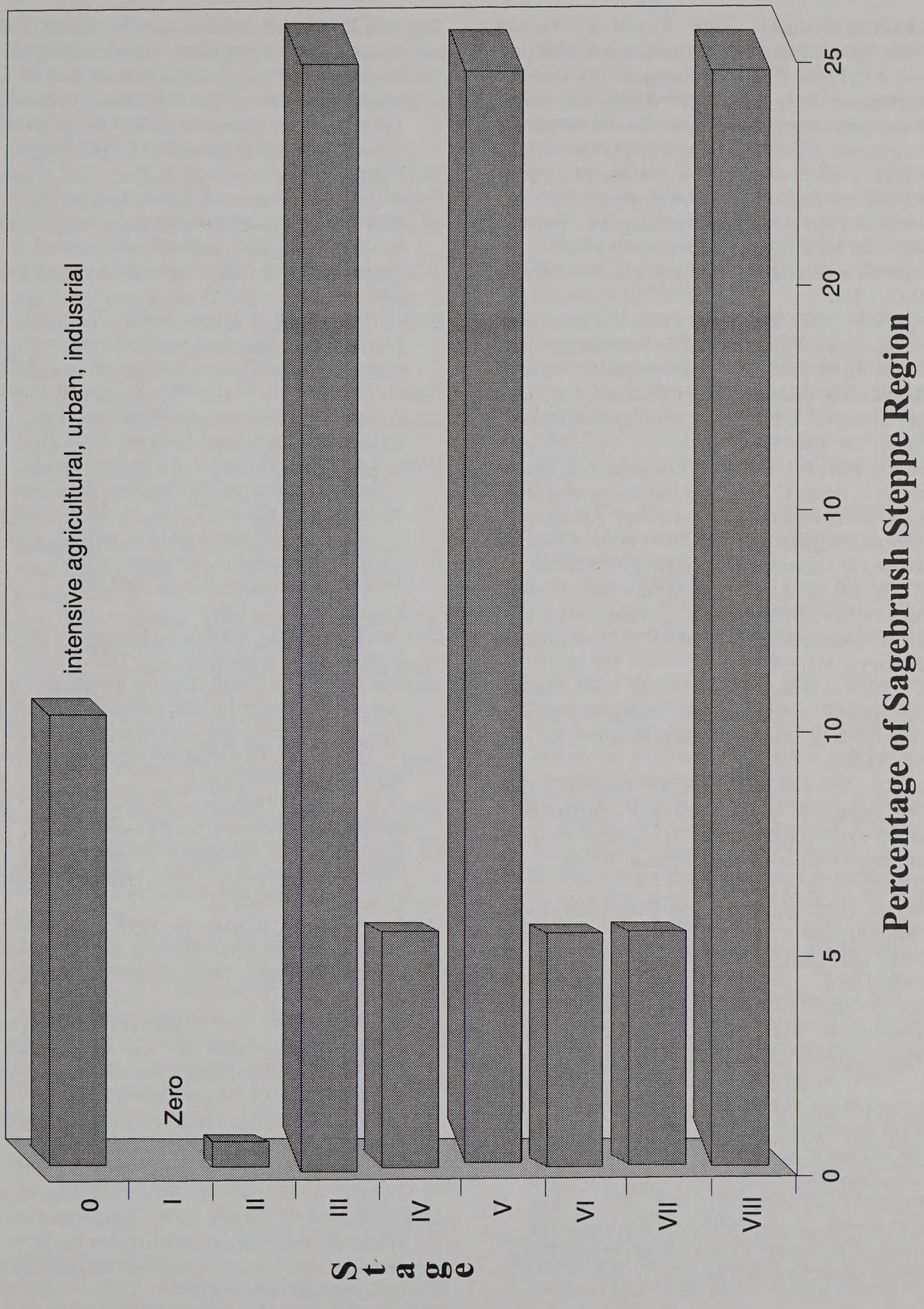

Figure 4. Percentages of the Pre-Columbian sagebrush steppe that are estimated to be occupied by the various states of Figure 3.

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SAGEBRUSH STEPPE WILDLIFE: HISTORICAL AND CURRENT PERSPECTIVES

Alan R. Sands
Signe Sather-Blair
Victoria Saab

INTRODUCTION

The sagebrush (*Artemisia* spp.) steppe ecosystem harbors about 250 species of terrestrial vertebrates, the majority being birds and mammals, with approximately 100 and 70 species, respectively (Braun et al. 1976). Many species that were formerly common and abundant now have restricted ranges separated by a vast landscape of agricultural developments and nonnative grasslands. While there are currently no federally listed wildlife species under the Endangered Species Act that would be considered sagebrush steppe obligates, some significant, formerly wide-ranging species such as the Columbian sharp-tailed grouse (*Tympananuchus phasianellus columbianus*) and sage grouse (*Centrocercus urophasianus*) (Washington State population only) have been petitioned for listing as threatened or endangered. A significant number of sagebrush steppe wildlife species are also identified as species of concern by federal land management and state wildlife agencies due to significant declines in distribution and abundance (Rich 1999). Nearly all declines of native sagebrush steppe vertebrates are closely associated with habitat loss or degradation.

Twenty-nine tall sagebrush communities and 14 short sagebrush communities have been described for the sagebrush steppe (Blaisdell et al. 1982). Precipitation, elevation, and soil conditions are major factors that influence the distribution of these communities. The structure and composition of plant species vary greatly within and among these communities (Daubenmire 1970, Franklin and Dyrness 1973, Hironaka et al. 1983, Anderson 1986). This heterogeneity creates a variety of ecological niches for wildlife (Dealy et al. 1981, Paige and Ritter 1999).

Wildlife habitat alterations (loss, degradation, and fragmentation) within the sagebrush steppe ecosystem have been and continue to be common and widespread. Major historical losses of sagebrush steppe occurred as a result of conversion to agricultural cropland, especially in eastern Washington and southern Idaho (Wisdom et al. In Press). During the middle decades of this century, millions of hectares were treated to convert sagebrush areas to nonnative grasslands for livestock forage production. More recently, extensive wildfires have converted millions of hectares to nonnative annual grasslands, especially in eastern Oregon, southern Idaho, and northern Utah and Nevada (Pellant and Hall 1994). As early as 1978, the combined effects of these historic alterations resulted in a 55% loss of the sagebrush steppe in Idaho (Sharp and Sanders 1978). Today, scientists estimate that sagebrush steppe habitat has been reduced by 1/3 in the interior Columbia River Basin ecoregion (Wisdom et al. In Press).

The patchwork of sagebrush areas remaining today is a landscape of habitat islands for sagebrush obligate species. Many remaining sagebrush communities are small and widely separated from each other. This habitat fragmentation has important implications to wildlife, especially those that are migratory and dependent on large sagebrush areas. For example, loss of low elevation sagebrush areas that provide crucial winter habitat for species such as sage grouse or mule deer (*Odocoileus hemionus*) can have a disproportionate effect on the population health of these species over a very large area (Swenson et al. 1987, Thomas and Irby 1990, Dobkin 1995).

Nearly all the remaining sagebrush steppe is ecologically degraded (West, this volume). Unregulated livestock grazing in the early 1900s resulted in a reduced herbaceous understory, subsequent decrease in the natural fire frequency, and a commensurate increase in sagebrush cover (Blaisdell et al. 1982, Young 1994). In some mesic sagebrush areas, this has also provided conditions suitable for expansion of conifers (*Juniperus* spp., *Pseudotsuga menziesii* and *Pinus* spp.) into sagebrush areas (West and Van Pelt 1987). In many areas, the reduction in native ground covers also created conditions suitable for nonnative annual grasses (Blaisdell et al. 1982, Young 1994).

These habitat alterations have caused considerable alarm among conservation biologists. Ungrazed shrub steppe has been recognized as a "critically endangered

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ecosystem" due to a loss of more than 98% from historical times (Noss et al. 1995). Similarly, the World Wildlife Fund ranked the Columbia Plateau and Wyoming Basin ecoregions, the 2 ecoregions encompassing most of the sagebrush steppe, as endangered and vulnerable, respectively (Ricketts et al. 1999). They considered the Columbia Plateau ecoregion as an area of very high biological importance. Habitat loss and significant threats of additional losses were the primary factors in their assessment. Although they did not rank the Wyoming Basin at the same level of biological importance or threat, the vulnerable rating was based on impending increase of energy and mineral development.

In this paper we briefly discuss the pristine habitat-wildlife conditions (defined as that which existed prior to or just after European settlement). We then summarize species-habitat relationships associated with the various ecological states of sagebrush steppe condition as defined by West (this volume). Fragmentation and other factors affecting wildlife populations and habitat are discussed. We recommend actions that should be initiated immediately to reverse the current trend of habitat loss and degradation.

PRISTINE VEGETATION, WILDLIFE ABUNDANCE AND DISTRIBUTION

Most range scientists agree that sagebrush steppe communities generally had a vigorous herbaceous layer of perennial grasses and forbs intermixed with a moderate sagebrush cover at the time of pre-European settlement (Franklin and Dyrness 1973, Harniss and Murray 1973, Vale 1975). In eastern Washington, Daubenmire (1970) found that relic big sagebrush (*Artemesia tridentata* ssp. *tridentata*)/bluebunch wheatgrass (*Pseudoroegneria spicata*) and big sagebrush/Idaho fescue (*Festuca idahoensis*) stands had an average canopy coverage of bluebunch wheatgrass and/or Idaho fescue of 45% and 58%, respectively; sagebrush canopy coverage averaged 14% (9 to 19%). After 25 years of grazing exclusion in a more xeric Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) site in eastern Idaho, Anderson and Holte (1981) reported the average basal coverage of perennial grasses at 6%, with total shrub canopy cover at 27%.

With a basic understanding of the pristine vegetative conditions and a knowledge of species-habitat relationships, some inferences can be made regarding historical wildlife habitat (Wisdom et al. In Press). Historical accounts from early explorers and pioneers are useful in reconstructing the original distribution of wildlife, but using them as descriptors of "natural" abundance should be avoided, especially for large mammals. Prior to European exploration and settlement, large mammal populations may have existed at levels below habitat potential due to the settlement distribution and influence of Native Americans (Martin and Szuter 1999). Trappers and early explorers in Nevada often noted a lack of large mammals, but they also frequently noted a high degree of shyness in game (R. McQuivey, Nevada Department of Wildlife, personal communication). This shyness is a behavioral attribute typical of species experiencing a high degree of persecution.

Based on vegetation and species-habitat relationships, large mammals such as elk (*Cervus elaphus*), pronghorn (*Antilocapra americana*), and bighorn sheep (*Ovis canadensis californiana*) should have been relatively common in pristine sagebrush steppe (Martin and Szuter 1999). Pronghorn are a diurnal species dependent on eyesight and speed to escape predators. Under pristine conditions, extensive areas with a low coverage of shrubs would have afforded good visibility for predator escape as well as adequate biomass to meet this species' food requirements (Yoakum 1980). Bighorn sheep are also dependent on keen eyesight to detect predators but use agility rather than speed for escape (Buechner 1960). Open sagebrush stands with good herbaceous understories near cliffs and other broken terrain offered very good habitat conditions for this species. Indeed, many of the relic sagebrush areas remaining today are associated with inaccessible canyonlands and other rugged habitats that are bighorn sheep source habitats.

In the Great Basin region, black-tailed jackrabbits (*Lepus californicus*) were the most abundant large herbivore (McAdoo and Young 1980, Wagner 1981). However, although jackrabbits were abundant during population peaks under pristine habitat conditions, they were never as abundant as they later came to be with the advent of farming and the increase in sagebrush resulting from livestock grazing (McAdoo and Young 1980).

Historically, sage grouse were widely distributed and abundant in many areas (Schroeder et al. 1999). The diversity of sagebrush cover and density of herbaceous ground cover must have provided ideal conditions. Many anecdotal references associated with the fall migrations of birds refer to hundreds and even thousands of birds (Patterson 1952). In 1886, naturalist G.B. Grinnell reported that literally thousands of birds passed by him one fall day in western Wyoming, reminding him of the flights of passenger pigeons (full quotation in Patterson 1952).

The mesic portions of the sagebrush steppe historically supported large numbers of Columbian sharp-tailed grouse (Connelly et al. 1998). Numerous reports from explorers and pioneers noted the high abundance of sharptails, even more so than sage grouse in some areas. Sharptails were frequently reported as the most abundant game bird in eastern Washington (Yocom 1952), eastern Idaho (Rust 1917), and northern Utah (Lee 1936).

WILDLIFE AND SAGEBRUSH STEPPE ECOLOGICAL STATES

Range ecologists recently have identified 8 ecological states for sagebrush steppe plant communities (West, this volume). These ecological states range from pristine

(State I) to annual grasslands (State VIII). We provide a brief overview of the native wildlife community within each of these ecological states, frequently comparing habitat conditions to pristine conditions (State I) as a frame of reference. Information concerning wildlife responses to these various states was often limited to birds and mammals; data on reptiles and amphibians are scarce.

Pristine (State I) and Relic (State II) Sagebrush Steppe

Pristine conditions likely no longer exist, and relic areas are thought to constitute less than 1% of the remaining sagebrush steppe habitat (West, this volume). Relic sites are areas characterized by open sagebrush stands with an abundant perennial herbaceous cover (West, this volume), similar to pristine conditions. The heterogeneous shrub-grassland habitats (Daubenmire 1970) provide generally good biological diversity (Dobler 1994) and diverse niches for shrub- and ground-nesting birds (McAdoo et al. 1986). Although these areas make up a small proportion of the landscape, they are usually associated with other sagebrush-dominated communities and are often important source habitats for sagebrush steppe species that prefer more open sagebrush cover (e.g., bighorn sheep, pronghorn, and sharp-tailed grouse).

Sagebrush with a Depleted Herbaceous Layer (State III)

Sagebrush areas with depleted understories occupy approximately 25% of the sagebrush steppe landscape (West, this volume). Wildlife preferring dense shrub cover (>20%) with little herbaceous understory for nesting or foraging would be favored in this ecological state. Habitat in this state may have a similar wildlife species richness as relic areas; but abundance, especially for ground-nesting birds, would likely be reduced (McAdoo et al. 1986).

Numerous studies on sage grouse have demonstrated the critical importance of sagebrush for both food and cover (Connelly and Braun 1997, Schroeder et al. 1999). Sagebrush cover is essential for nesting and wintering habitats, characterized by average canopies between 10% and 30%. However, it is becoming increasingly clear that a vegetatively diverse sagebrush community with native perennial understory may provide the best habitat for nesting sage grouse (Apa 1998, Schroeder et al. 1999).

Most remaining sage grouse habitat is in this ecological state with varying degrees of understory depletion. Poor nesting habitat conditions have been a documented wildlife management concern for many years (Patterson 1952, Autenrieth 1981). Nest predation rates have been reported as significantly higher in sagebrush stands with a depleted perennial herbaceous layer (Connelly et al. 1991, Gregg et al. 1994, Delong et al. 1995, Sveum et al. 1998).

Sage thrashers (*Oreoscoptes montanus*) showed a positive relationship to an understory of bluebunch wheatgrass (Dobler 1994). Other ground-nesting birds such as vesper sparrow (*Pooecetes gramineus*) and western meadowlark (*Sturnella neglecta*) occur at much lower densities in sagebrush stands with a depleted native herbaceous understory (Wiens and Rotenberry 1981, Petersen and Best 1987).

The percent sagebrush cover has important influences on habitat use by many bird species. Of 17 birds studied in eastern Washington, 7 species had a positive relationship to sagebrush cover, 2 were inversely related, and 8 were not related (Dobler 1994). Species benefiting from sagebrush cover included Brewer's sparrow (*Spizella breweri*), sage sparrow (*Amphispiza belli*), sage thrasher, loggerhead shrike (*Lanius ludovicianus*), brown-headed cowbird (*Molothrus ater*), and mourning dove (*Zenaida macroura*). Species with a negative relationship were savannah sparrow (*Passerculus sandwichensis*) and long-billed curlew (*Numenius americanus*).

Black-tailed jackrabbits, strongly dependent on shrubs, have expanded their range, whereas in contrast, the distribution of white-tailed jackrabbits (*Lepus townsendii*), a species more dependent on grass, has diminished (McAdoo and Young 1980). However, on a finer scale, black-tailed jackrabbit populations have been significantly reduced where wildfire has eliminated sagebrush stands in the Snake River Plain (USDI 1996, Knick and Dyer 1997).

Native Perennial Herb-Dominated Stands (State IV)

This state is considered transitional and occurs after burns or other shrub-removal treatments. Less than 5% of the sagebrush steppe is in this state (West, this volume). Grassland bird species such as vesper sparrow, western meadowlark (Wiens and Rotenberry 1981, Castrale 1982), and sharp-tailed grouse (McDonald 1998) are favored in this state, although its value to wildlife depends on local conditions often related to the intensity and timing of livestock use (Saab et al. 1995)

No differences were reported in total density or biomass of songbirds following a sagebrush fire, although species composition changed dramatically (Wiens and Rotenberry 1981). Horned lark (*Eremophila alpetris*) replaced sage sparrow as the most abundant breeding bird. Nongame bird species richness and abundance increased 4 years after a mosaic-pattern prescribed burn in eastern Idaho (Peterson and Best 1987). One ground nester increased in abundance, and 2 species colonized the burn areas. However, a similar prescribed burn in the same region has resulted in the continued depression of a nesting sagegrouse population 9 years after the burn (Connelly et al. 1994).

Sagebrush with an Annual Herbaceous Layer (State V)

Approximately 25% of the sagebrush steppe landscape is now thought to be occupied by sagebrush with an understory dominated by nonnative annual grasses (West, this volume). Sagebrush communities in this state are extremely vulnerable to loss and permanent transition to State VIII as a result of wildfires (Pellant 1990; Shaw et al. 1999; West, this volume).

Sagebrush canopy cover and structure may be similar to State III areas, but herbaceous conditions are significantly different. Understory vegetation of nonnatives provides marginal nesting cover for ground-nesting birds such as sage grouse, sharp-tailed grouse, vesper sparrow, and western meadowlark (McAdoo et al. 1986, Saab and Marks 1992, Dobler 1994, Saab and Rich 1997). Nesting conditions for these species are particularly adverse during drought periods when annual grasses would provide only limited concealment. Four of 7 bird species studied in shrub steppe habitats of eastern Washington showed an inverse relationship to annual grass cover, and no species showed a positive relationship (Dobler 1994). The most common sagebrush obligates found in these sites are shrub nesters, including Brewer's sparrow, sage thrasher, and sage sparrow (Dobler 1994, Knick and Rotenberry 1995).

Introduced Perennial Grass (State VI)

Approximately 5% of the sagebrush steppe landscape is now in this state due to fire rehabilitation efforts and land treatments for forage production. Until recently, rehabilitation efforts of degraded rangelands largely involved the use of nonnative perennial grasses, usually as a single species. The most widely used grass has been crested wheatgrass (*Agropyron cristatum*), although intermediate wheatgrass (*A. intermedium*) has been used extensively in mesic sagebrush steppe sites (generally >30.5 cm [12 inches] annual precipitation). Seedings within the past 10 to 15 years usually involved multiple herbaceous species. Although some seedings included native grasses and shrubs, the use of native shrubs, grasses, and forbs is still quite limited.

Few if any wildlife studies have been done on multiple-species seedings. Studies have shown that single-species nonnative grasslands provide poor habitat for native sagebrush steppe birds. Nesting western meadowlarks and vesper sparrows were more abundant in native perennial grasses than in crested wheatgrass seedings (Wiens and Rotenberry 1981). Total bird species density, richness, and diversity in crested wheatgrass stands in southeastern Idaho were lower than in nearby sagebrush habitats (Reynolds and Trost 1980). Horned larks (*Eremophila alpestris*), western meadowlarks, and vesper sparrows were found nesting in ungrazed crested wheatgrass seedings, but nothing is known about their reproductive success.

Seedings may function as habitat sinks, where mortality exceeds reproduction (cf. Saab and Rich 1997). In eastern Washington, Columbian sharp-tailed grouse selected crested wheatgrass for nesting. Their nest success, however, was only 18% (n=11), whereas nest success in native grass and shrub habitats was 100% (n=6) (McDonald 1998). McDonald (1998) considered crested wheatgrass seedings to be habitat sinks for sharptails and recommended their replacement with native bunchgrass and forb species.

Other nonnative grasses may not provide good habitat for sharptails. Columbian sharp-tailed grouse in western Idaho avoided use of an intermediate wheatgrass seeding within their home range (Marks and Marks 1987, Saab and Marks 1992). Additionally, native perennials such as bluebunch wheatgrass and arrowleaf balsamroot (*Balsamorhiza sagittata*) were highly selected cover species during a drought year (Saab and Marks 1992).

Ungrazed nonnative grasslands seeded through the federal Conservation Reserve Program have provided nesting and brood-rearing habitat for Columbian sharp-tailed grouse in southeastern Idaho (Idaho Department of Fish and Game, unpublished data). However, seedings containing dryland alfalfa (*Medicago sativa*) or with an abundance of annual forbs had greater use by sharptails than seedings that were predominantly grasses (Ulliman 1995).

Numbers of small mammals and reptiles also have been reduced in nonnative seedings. In southeast Idaho, lower rodent and reptile densities were found in crested wheatgrass seedings compared to sagebrush stands (Reynolds and Trost 1980).

Sagebrush with an Introduced Perennial Grass Understory (State VII)

Approximately 5% of the sagebrush steppe landscape is thought to be older nonnative grass seedings with some sagebrush (West, this volume). Little data are available on responses of wildlife to sagebrush re-establishment into these areas. In a central Nevada study, species richness was greater where sagebrush had established into crested wheatgrass seedings than in either monoculture seedings or high-coverage sagebrush habitats (McAdoo et al. 1986). Comparable levels of abundance may not occur, however, unless microhabitat structure is similar to that of native plant species (see discussion in previous section, State VI). The ecologically simpler habitat is likely to have a lower wildlife diversity than sagebrush with an understory of native grass and forb species.

Annual Grasslands (Type VIII)

Annual grasslands now occupy more than 25% of the sagebrush steppe landscape, and this statistic is growing (West, this volume). Most sagebrush steppe species have not benefited from the loss of shrubs and the dominance of annuals (Dobler 1994). Shrub obligate species such as Brewer's sparrows and sage grouse largely disappear from previously occupied areas. Direct impacts to shrub-nesting species occur with the loss of nesting and foraging substrates. Some species of non-game birds that are not dependent on shrubs for nesting either decline or are eliminated by the loss of shrub cover.

In south-central Idaho, songbird community composition and density were dramatically altered in cheatgrass (*Bromus tectorum*) compared to native sagebrush cover (T. Rich, unpublished data in Shaw et al. 1999). From 1981 to 1985, species richness averaged 8.4 to 10.2 in sagebrush stands while nearby cheatgrass stands averaged 1.5. Breeding densities were also strikingly reduced in cheatgrass stands. Densities averaged 0.6 to 1.1 birds/ha in cheatgrass compared to 3.9 to 8.1 birds/ha in sagebrush. Small mammal populations in a cheatgrass-dominated rangeland in Washington were only 1/3 as abundant as those on adjacent sagebrush/bitterbrush (*Purshia tridentata*)-dominated sites (Gano and Rickard 1982).

Studies in the Snake River Birds of Prey National Conservation Area (NCA) in southwest Idaho suggest that golden eagles (*Aquila chrysaetos*) and prairie falcons (*Falco mexicanus*) in the NCA have been adversely affected by changes in prey species abundance as a result of annual grassland expansion and corresponding loss of sagebrush cover (USDI 1996, Marzluff et al. 1997, Kochert et al. 1999, Steenhof et al. 1999). Black-tailed jackrabbit population declines were closely correlated with a loss of sagebrush cover, and current distribution was related to remaining habitat (USDI 1996, Knick and Dyer 1997). Densities of Paiute ground squirrels (*Spermophilus mollis*) (formerly Townsend's ground squirrels [*Spermophilus townsendii*]) in the same area could be high even with the loss of sagebrush cover and dominance of annual grasses. Researchers, however, found that squirrel populations fluctuated more dramatically in areas that had been converted to annuals. Populations were more stable in sagebrush communities with a residual component of native herbaceous perennials in the understory (USDI 1996).

At least 2 bird species have apparently benefited from the expansion of annual grasslands. Long-term breeding-bird census data indicate that long-billed curlews and western burrowing owls (*Speotyto cunicularia*) have increased in recent decades (Saab and Rich 1997, Wisdom et al. In Press). Cheatgrass and medusahead (*Taeniatherum caput-medusae*) are suitable for these species because they favor open habitats with short vegetation. However, these population gains may be short-lived. The rapid replacement of cheatgrass- and medusa-infested ranges with taller exotic annual forbs may render these sites unsuitable to these species (Shaw et al. 1999).

Annual Forb-Dominated Stands (Proposed State IX)

Large areas are now becoming dominated by exotic annual forbs such as yellow starthistle (*Centaurea solstitialis*), knapweeds (*Centaurea* spp.), rush skeletonweed (*Chondrilla juncea*), and other exotics. Perhaps we are beginning to see yet another sagebrush steppe state that represents a greater magnitude of degradation for this ecosystem and its associated wildlife.

The consequences of transformations to this state of ecological degradation are largely unstudied, but the implications are particularly onerous to wildlife.

INFLUENCES OF HABITAT FRAGMENTATION

Sagebrush patch sizes, surrounding landscapes, and connectivity of suitable habitats are critical to the long-term persistence of many sagebrush steppe species. Habitat fragmentation and patch sizes may influence wildlife use and productivity as much as microhabitat conditions (Knick and Rotenberry 1995). For example, sagebrush patch sizes influenced black-tailed jackrabbit distribution in the Snake River Birds of Prey Area (Knick and Dyer 1997), which in turn affected the distribution, habitat use, and productivity of golden eagles (Marzluff et al. 1997, Kochert et al. 1999).

Some sagebrush steppe species require thousands of hectares to support viable populations. These area-sensitive species include both large and medium-sized mammals and birds such as sage grouse and sharp-tailed grouse. This is not to imply that their habitat must be either all in relic condition or all in sagebrush habitats. Indeed, some species like mule deer and black-tailed jackrabbits may flourish in moderately degraded habitat, and others such as pronghorn may sustain low-density populations in annual grasslands.

Within the interior Columbia Basin, major loss and fragmentation of sage grouse habitat has occurred since settlement (Fig. 1) (Wisdom et al. In Press). Similarly, landscape analysis of historic and current Columbian sharptail habitat in eastern Washington revealed that their habitat has declined 83%, while their distribution has decreased 89% (McDonald and Reese 1998). Additionally, the number of habitat patches nearly doubled and mean habitat patch size declined 36%, from 4,474 ha (11,051 acres) to 2,857 ha (7,057 acres) (McDonald and Reese 1998). As a result of fragmentation, the mean distance between populations is currently 61 km (38 miles), triple the dispersal distance of female sharptails.

Sage grouse and sharp-tailed grouse need thousands of hectares of adequately connected habitat to support self-sustaining populations (Paige and Ritter 1999). An estimated 3,000 ha (7,400 acres) are needed to support a population of sharptails, with at least 33% of the area undisturbed habitat imbedded within other habitats that provide some value to the species (Connelly et al. 1998). Sage grouse, with their narrower habitat requirements and virtual dependence on sagebrush, are likely to require larger and more continuous sagebrush habitats than sharptails.

Within the Snake River Birds of Prey National Conservation Area, fragmentation of shrub steppe significantly influenced the presence of shrub-obligate species (Knick and Rotenberry 1995). They found that sage sparrows, Brewer's sparrows, and sage thrashers were all sensitive to the amount of shrub cover and the

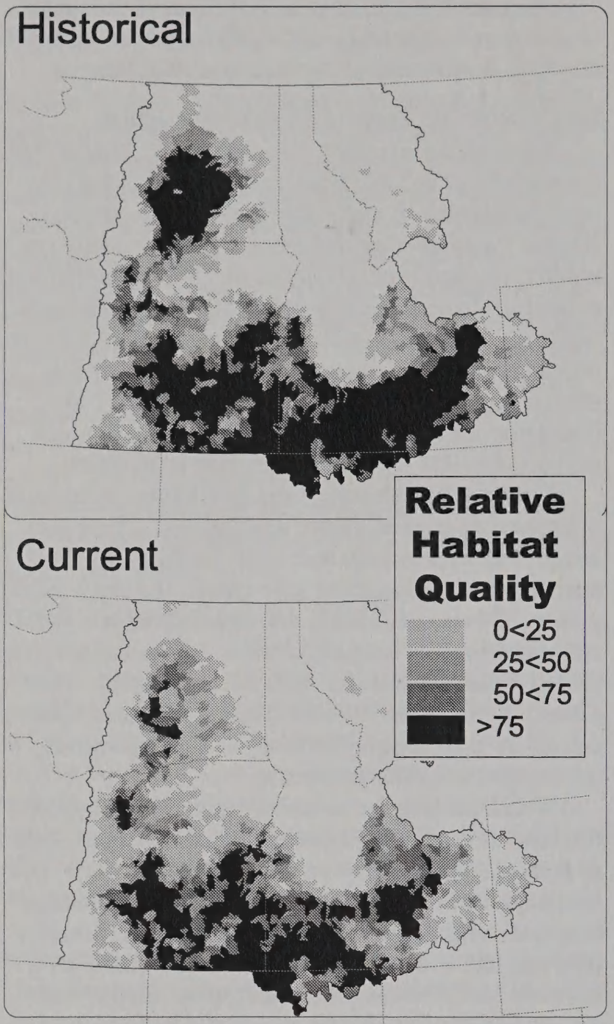

Figure 1. Historical and current sage grouse habitat, Interior Columbia River Basin (Wisdom et al. In Press).

shrub patch size. In a Washington study, sage sparrows did not occur on shrub patches less than 130 ha (320 ac) (Vander Haegan, personal communication, in Paige and Ritter 1999). To support a population, a much larger area of suitable habitat would be needed.

CONCLUSIONS

Studies of wildlife in sagebrush communities have shown consistent patterns. Ecologically intact sagebrush communities have a higher diversity of species than degraded sites (Petersen and Best 1987, Dobler 1994). Moreover, most species that are currently rare or have undergone significant declines are closely associated with sagebrush communities that are still ecologically intact (i.e., retain characteristics of unaltered sagebrush communities).

We have at least rudimentary knowledge of species/habitat relationships for many sagebrush-associated wildlife species and the range of natural variation in sagebrush vegetative communities. We know sagebrush communities are very vulnerable to degradation and are difficult, if not impossible, to restore once certain thresholds are crossed (West, this volume). Lastly, we know that altered areas are vulnerable to further degradation, providing even less habitat for sagebrush steppe wildlife.

MANAGEMENT IMPLICATIONS

Our goal is to maintain native species biodiversity, referring to both the number of species as well as an intrinsic level of abundance that provides for long-term population persistence in the presence of expected environmental perturbations (e.g., flood, fire, and drought). Providing for the needs of area-sensitive species dependent on intact sagebrush steppe communities should provide for the needs of many other sagebrush steppe-associated species. To accomplish this, we must approach management from a landscape perspective, even though specific management actions are nearly always implemented at the local level. Local decisions should consider the landscape context when implementing management.

Assuming there will continue to be limited financial resources to accomplish wildlife diversity goals, the following recommendations are listed in priority order:

1. Identify and maintain the ecological integrity of remaining intact sagebrush steppe communities. The investment in time and resources is minimal to accomplish this while these areas are still intact.
2. Identify areas that are depleted (States III, IV, and VI) but can be restored using "soft" energy inputs (see West, this volume). Implement management actions to protect and recover these sites. Use adaptive management to monitor progress and make appropriate changes in management strategies.
3. Identify areas that are severely degraded (States V - VIII) but are key to reconnecting fragmented habitats. Restore these areas using native plants (shrubs, grasses, and forbs) as available. If nonnatives are selected, use ecotypes that closely mimic growth forms of native species. Manage land uses to maintain restored habitats and protect financial investment.

In addition, achieving additional understanding and support for sagebrush steppe conservation and accelerating applied native plant research efforts to more effectively restore depleted habitats are essential to a successful program to maintain a viable sagebrush steppe ecosystem.

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HISTORICAL SAGEBRUSH ECOSYSTEMS: HUMAN INFLUENCES

James A. Young

Introduction

The conversion of millions of hectares of sagebrush (*Artemisia*)/bunchgrass rangelands to dominance by the accidentally introduced, self-invasive annual cheatgrass (*Bromus tectorum*) is a well-documented fact. The purpose of this paper is to provide a historical perspective on how the scientific community first perceived this conversion.

Social, Ecological, Economic Setting

The initial establishment and rapid spread of cheatgrass in the Intermountain Region largely occurred during the 1900s, with increasing dominance during the second half of the century. Until 1934, the portions of the sagebrush/bunchgrass rangelands that were not in private ownership or within a National Forest were publicly owned land open to homesteading, with no grazing management. After passage of the 1934 Taylor Grazing Act, these lands were administered by the U.S. Department of the Interior (USDI) Grazing Service and later by the USDI Bureau of Land Management. The public lands outside of the National Forests were generally the lower-elevation areas with less potential for plant growth. These lands included homesteaded areas where cropping had failed and the land was subsequently abandoned.

For the first 3 1/2 decades of this century, these vacant public lands were grazed in common by domestic cattle, horses, and sheep. Huge numbers of draft horses were turned loose on the open range during the off season for agricultural production. This was especially true near newly irrigated agricultural developments such as on Idaho's Snake River Plain. It is very difficult now to visualize and ascertain the biological impact of range sheep in the sagebrush/bunchgrass ecosystem during the first half of the 20th century. The sheep industry grew after the cattle and horse husbandry industries were already established (Young and Sparks 1985). The enterprises that owned no base property and were known as "tramp sheep" contributed to the destruction of range resources. This huge industry was superimposed upon already overstocked rangelands. Ranchers herded their cattle on sagebrush ranges to make sure that no forage went ungrazed, because if they did not utilize the resource, their neighbor – or worse yet, a "tramp sheep" outfit – would get the forage (Emmerich et al. 1992).

If you continue to excessively graze sagebrush/bunchgrass ranges season-long, 2 things will happen. First, the perennial grasses will disappear, and second, the density of big sagebrush will increase. The second is a forgotten factor in modern big sagebrush management. With virtually no herbaceous understory to help carry wildfires, the overly dense big sagebrush stands perpetuated themselves while limiting the establishment of native herbaceous perennials.

Introduction of Exotic Invasive Weeds

Starting with Russian thistle (*Salsola targus*), tumblemustard (*Sisymbrium altissimum*), cheatgrass, medusahead (*Taeniatherum caput-medusae*), and barbwire Russian thistle (*Salsola paulsenii*), the exotic self-invasive species have come in waves. They form a seral continuum that has truncated succession and led to long-term dominance by cheatgrass. The factor that invokes this dominance is stand renewal by repeated wildfires.

The truncation of succession and the relation of cheatgrass dominance to rapidly recurring wildfires was first reported by Pickford (1932). His classic paper on the spring-fall ranges of Utah dramatically reported what eventually would happen to much of the sagebrush/bunchgrass ranges of the Intermountain Region.

Grazing Management with Cheatgrass

Stewart and Young (1939) reported that the short "green-feed" period, great variability among years in herbage production, and potentially injurious awns made cheatgrass a hazardous species on which to base livestock production. Aldo Leopold (1941) followed with a paper stressing that the increased chance of ignition and rate of spread of wildfires fueled by cheatgrass would prove very harmful to wildlife populations. Before the Grazing Service was established, utilization of forage on the sagebrush ranges was generally so intense that cheatgrass was apparently biologically suppressed. Fleming et al. (1942) published a landmark bulletin on the value of grazing *Bromus tectorum* (they called it bronco grass). They readily admitted it was an inferior forage to the disappearing native bunchgrasses, but reality said it was the forage that supported a significant portion of the range livestock industry.

The final landmark scientific paper on cheatgrass was published by Robertson and Pearse (1945). Their premise was that cheatgrass, by out-competing the

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seedlings of artificially planted perennial grasses for soil moisture, was virtually closing communities to the recruitment of new perennials. Eventually it became apparent that their findings extended to virtually all perennial seedlings, not just introduced forage grasses.

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Current Issues in Sagebrush Steppe Ecosystems

CURRENT ISSUES IN THE SAGEBRUSH STEPPE ECOSYSTEM: GRAZING, FIRE, AND OTHER DISTURBANCES

Lee E. Eddleman
Paul S. Doescher

Disturbances under consideration are those that result, either directly or indirectly, from human activities in the sagebrush steppe ecosystem. Grazing by livestock as well as fire use and removal are of primary concern, although other direct activities such as recreation will be touched upon.

By definition, disturbance means a significant change has occurred in the resource base, the plant community has been moved away from a stable state, and a compositional change has occurred in both plant species and life histories. Key functional elements of any disturbance are its timing (seasonality), intensity (resource loss), abiotic resources available (water and nutrients), biotic resources available (species and their attributes), frequency (recovery interval between disturbances), and regime (connectivity to other disturbances in time and space) (Sousa 1984).

Issues surrounding grazing and fire tend to arise out of the ecological uncertainty as to whether they will produce a feedback that enforces the stability of the present community or whether they will promote transitions to a more desired community or a less desired one. Given the present state of the sagebrush steppe ecosystem, key questions center on how to influence sagebrush communities through the presence or absence of grazing and fire. The effects on vegetation and soils from overgrazing, high-frequency fires, and other factors such as uncontrolled recreational vehicle use may be rather obvious (Blaisdell et al. 1982, Bunting et al. 1987, Vavra et al. 1994). Less obvious, however, are the effects on other biota. Judicious grazing practices and prescribed fire carry with them varying degrees of uncertainty as to short-term and long-term outcomes. This degree of uncertainty can be expected, since the key functional elements of disturbance vary greatly through time. Further, in the presence of a highly variable climate, they function as a disturbance regime rather than as independent events.

In the sagebrush steppe communities of the Intermountain West, original plant communities were composed of a few dominant species, i.e., sagebrush and 1 or 2 perennial grasses, and numerous other species that were both spatially and temporally dynamic. Those few dominant species were highly competitive for limited resources and tended to produce a relatively stable sagebrush steppe ecosystem in the face of a variable disturbance regime. Their relative abundance locally and regionally was mediated largely by fire, herbivory, and climate. The abundance of the dynamic group, comprising the vast majority of sagebrush steppe species, including many forbs, was mediated by disturbances that freed up resources for establishment. Most of these species relied on seed production and dispersal as a means of maintaining their presence in the system and establishment on disturbed sites.

Current human activities, i.e., grazing, fire, and recreation, in the sagebrush steppe are not perpetuating the original plant community composition. West (1999) estimated that less than 1% of the sagebrush steppe remains in its original condition. Rather, we have a system in which disturbances cause several very different changes in species composition to occur. First, disturbance may enhance the competitive ability of one of the dominant species, i.e., sagebrush, and reduce the competitive ability of the other dominant species, i.e., perennial grass. Second, disturbance may enhance the competitive ability of one dominant, i.e., sagebrush, and eliminate the other dominant, i.e., perennial grass. Third, disturbance may cause the loss of the original dominants. In all 3 cases, one or all of the original dominants are required to function in the ecosystem similarly to the dynamic disturbance-adapted species such as cheatgrass, which they are not well adapted to do.

West (1999) used the state-and-transition model to describe current conditions in the sagebrush steppe. He recognizes 8 states that range from pristine to highly disturbed. Four of these, which we would place in the moderate to highly disturbed state, make up the vast majority of the sagebrush steppe. These 4 states, using West's model (included in this proceeding), are "*II: Late Seral Sagebrush Steppe*," "*III: Depauperate Late Seral Sagebrush Steppe*," "*V: Brush with only introduced*

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annuals in understory," and "*VII: Cheatgrass/Medusahead*." States III and V together constitute over 50% of the sagebrush ecosystem. States II and VII make up most of the rest of the sagebrush steppe ecosystem.

At the heart of management are the issues of prescribed grazing and fire and their effects on transitions toward desired communities and, conversely, their effects on transitions to less desirable states. Key questions that need to be asked for each and every vegetation state are:

- What site resources are available?
- What transitions and steady states are possible?
- How do grazing and fire direct plant succession?

Those questions should form the basis of management decisions before they are implemented.

Those plant communities falling into the state *Late Seral Sagebrush Steppe* should be considered for maintenance of the vegetation community. Even here, perceptions of biodiversity and health may push managers to consider activities that lead to improvement and, conversely, the elimination of activities that lead to less desirable communities. *Late Seral Sagebrush Steppe* communities likely contain a good abiotic and biotic resource base to work from. While prescribed fire may be used to temporally increase the dominance of bunchgrasses, only carefully managed grazing will prevent a transition to the *Depauperate Late Seral Sagebrush Steppe*.

Depauperate Late Seral Sagebrush Steppe communities are in the most critical state. Site resources, including the dominant bunchgrasses, are present but limited in abundance. Grazing and fire have the potential to cause transitions to one of several other steady states. A key question to be asked of these communities is, "Will any kind of prescribed fire lead to an increase in bunchgrasses?" Also of great concern is the question as to whether there is any way to manage grazing to increase the abundance of bunchgrasses. Concern that grazing of any kind may cause the transition across the successional threshold to the less desirable state, *Brush with Introduced Annual Understory*, is certainly justified.

Sagebrush steppe communities in the *Depauperate Late Seral Sagebrush Steppe* are very susceptible to being replaced by less desirable states dominated by shrub species and introduced weedy species. Once the threshold has been crossed to states that no longer contain the original dominant bunchgrasses, grazing and fire by themselves have lost their potential as effective tools for restoration. Only with the artificial addition and manipulation of available site resources through such practices as seeding, use of herbicides, etc., do they regain their potential as effective tools.

It may be well to remember that the sagebrush steppe functions well in the presence of a disturbance regime and that prescriptions for fire alone or grazing alone are much less likely to be successful than prescriptions inclusive of fire and grazing placed into the context of drought. The appropriateness of carefully considering the impacts of disturbance regimes on future plant community composition seems most critical for *Depauperate Late Seral Sagebrush Steppe* communities.

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INVASIVE EXOTIC PLANTS IN SAGEBRUSH ECOSYSTEMS OF THE INTERMOUNTAIN WEST

David A. Pyke

ABSTRACT

The most pervasive weeds of the sagebrush steppe, namely cheatgrass (*Bromus tectorum)* and medusahead (*Taeniatherum caput-medusae*), pose the greatest immediate threat for converting this vast ecosystem into a near monoculture of exotic annual grasses. Only 3 states (California, Oregon, and Utah) list medusahead as a noxious weed. Most states have elected not to place these species on their state noxious weed lists. The current distribution of cheatgrass far exceeds that of medusahead. Surveys show that cheatgrass occurs or has the potential to occur throughout the sagebrush steppe. Medusahead has become a major problem on clay soils in Oregon, southern Idaho, northern California, northeastern Nevada, and isolated locations in Washington. Although medusahead has not been collected or seen in Montana, this state should heed the experiences of Nevada and Utah, where recent discoveries remind us that this species is continuing to expand its range. Diverse, undisturbed environments do not always protect sites from invasions. Plastic seed production allows populations to maintain themselves in poor years and to increase in good years. These annuals often expand into the interspaces between native plants that were once occupied by biological soil crusts. They also produce large amounts of litter that decompose slowly, thus providing a site for their own seed banks to build and for wildfire fuel. Unfortunately, many areas of the sagebrush steppe have not seen the end of weed invasions. A survey of the literature revealed at least 46 exotic species that are commonly viewed as weeds and are capable of sustaining populations in sagebrush ecosystems. Of these, I classified 20 species as highly invasive and competitive. They possess traits that may allow them to successfully establish and sustain viable populations should seeds be introduced into diverse native sagebrush communities, even without human-caused disturbances. Managers should take precautions to halt the further spread of these species on their lands.

INTRODUCTION

The genus *Artemisia* L. (sagebrush) is estimated to have once occupied between 39 and 57 million ha of land in the Intermountain West of the USA (Tisdale et al. 1969, Chadwick 1989). Much of this land is described as semiarid and is dominated by one of four subspecies of big sagebrush (*A. tridentata*) (Shultz 1986, Rosentreter and Kelsey 1991). West (1983*a*,*b*) describes two major sagebrush ecosystems that occur in the Intermountain West. In the northern portion of the region, the plant communities exhibit a shared dominance between sagebrush and perennial grasses. The plant communities in the southern portion are dominated by sagebrush, with herbaceous species forming a subdominant role.

Before European settlement, fire was an important environmental (lightning-caused) and human-induced (Native American-caused) force that temporarily drove these ecosystems toward perennial grass dominance. During the intervals between fires, succession allowed shrub recovery. Fires would typically occur every 20 to 100 years, with intervals being shorter in the wetter, more productive mountain big sagebrush (*A. t.* ssp. *vaseyana*) communities and longer in the drier Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) communities (Miller et al. 1994).

The advent of European settlement in the Intermountain West began in the mid-1800s. During the first 60 years, a combination of overgrazing by livestock and introductions of competitive exotic plants set the stage for dramatic changes in plant communities (Miller et al. 1994). Invasive exotic plants, such as cheatgrass, spread quickly across the Intermountain West during the first 100 years after European settlement (Mack 1981, 1986). On Bureau of Land Management (BLM) lands in the Intermountain West, cheatgrass and medusahead now dominate or threaten to dominate over 30 million ha (Pellant and Hall 1994).

Some invasive exotic plants are so common in the Intermountain West that many states do not include these species on their noxious weed lists (Table 1). In sagebrush communities, exotic annual grasses provide sufficient fine fuels to reduce the fire-return intervals and eliminate fire-sensitive native shrubs (West 1983*a*,*b*). Although these annual grasses have made significant changes to sagebrush ecosystems, there is no guarantee

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that the annual grass communities they became have stabilized. Other invasive exotic plants such as yellow starthistle (*Centaurea solstitialis*) may replace the annual grasses in some locations (Sheley et al. 1999).

In the U.S. National Strategy for Invasive Plant Management, the first national goal is the effective prevention of the spread of invasive plants (FICMNEW 1997). To achieve this goal, people must understand and educate others on how invasive plants spread and establish in ecosystems. In this paper, I will provide an overview of the exotic plants with disturbance or invasive traits that either currently exist in or have the potential to invade these sagebrush ecosystems. I will examine traits and mechanisms of invasive exotic annual grasses that allow invasion and dominance in sagebrush ecosystems.

OVERVIEW OF EXOTIC INVASIVE PLANTS IN SAGEBRUSH ECOSYSTEMS

The influx of Europeans into the Intermountain West in the middle 1800s soon brought introductions of European plants. Settlement of the region required the production of agricultural commodities for subsistence and exchange among settlers. Miller et al. (1994) illustrated the fortuitous nature of the timing of settlement in this region. The Little Ice Age ended in the middle 1880s. The climate began to warm and precipitation was above normal. Settlement occurred between two major drought cycles, the 1840s and 1930s. Therefore, farming and ranching practices in the region were likely more successful during this period than if they had begun at another time. This success led to expansion of farming throughout the region.

Many of the early introductions of invasive exotic plants into the region occurred through crop seed contamination or through attachment on or ingestion and defecation by livestock. During the late 1800s, unplowed sagebrush lands had collections or notations of exotic invasive plants such as quackgrass (*Elytrigia repens* var. *repens*), redstem filaree (*Erodium cicutarium*), black mustard (*Brassica nigra*), rape mustard (*Brassica rapa*), shepherd's purse (*Capsella bursapastoris*), lambsquarters (*Chenopodium album*), horehound (*Marrubium vulgare*), cowcockle (*Vaccaria hispanica*), and several annual *Bromus* species, including cheatgrass (Mack 1986, Yensen 1981).

Livestock numbers in the Intermountain West peaked in the early 1900s (Young et al. 1976). Not only were the densities of animals high, but they were present throughout the entire year. This led to widespread overgrazing throughout the region (Griffiths 1902). As forage became scarce, livestock managers even set fires to eliminate fire-sensitive shrubs such as big sagebrush in the hopes of increasing the herbaceous component (Pechanec and Hull 1945). The early surveys of range condition noted increases in exotic annual *Bromus* species in several locations; however, the suspected problem species were soft brome (*B. mollis*) and cheat (*B. secalinus*) (see references in Mack 1986).

By the drought of the 1930s, the sagebrush ecosystems had undergone numerous introductions of exotic and decreases of native species in many locations. Beginning as early as the 1920s, many dry-land farmers in the region went bankrupt and abandoned their farms (Yensen 1981). The void left by land abandonment and by overgrazing was quickly filled by common ruderal species from Europe and Asia that originally arrived with the crop seeds and were often spread through the distribution of that seed. Russian thistle (*Salsola kali* ssp. *tragus*), flixweed (*Descurainia sophia*), and tumblemustard (*Sisymbrium altissimum*) became prominent during this time period. The enactment of state and federal seed laws, such as the Federal Seed Act of 1939, helped reduce the transport and spread of exotic species listed as noxious weeds. Many species, like cheatgrass, had already reached their current range by the time these laws were passed; but for others with only isolated introductions, these laws have no doubt slowed their spread.

Many of the exotic plants found in sagebrush ecosystems require continued disturbance of the soils or plant community to sustain their existence. Halting the disturbance and allowing recovery of the native vegetation is often all that is necessary to reduce or eliminate many of the exotics. In Table 1, I suggest that the non-bold species require some form of disturbance to maintain their dominance within sagebrush ecosystems. Plants like halogeton (*Halogeton glomeratus*) and Russian thistle fit this category. They both require disturbance to maintain dominance, but recovery of native plants or revegetation with desirable plants provides the necessary competition or changes in nutrient status to shift the dominance away from the exotic plant (McLendon and Redente 1991, Whitson et al. 1996).

For other exotics, disturbances such as fire may stimulate germination from the seed bank or cause heavy reproduction immediately after fire, thus allowing them to become temporarily prominent in the community. Examples of species in this category include some of the annual mustards, such as flixweed and tumblemustard. Heavily grazed and trampled locations may also favor some species, e.g., bur buttercup (*Ranunculus testiculatus*).

I consider those species shown in Table 1 in bold as highly invasive and capable of dominating a site once they are introduced. Within sagebrush ecosystems, we have several species that fit each of the three phases of the invasion process: introduction, colonization, and naturalization (Groves 1986). Establishment and maintenance of these species in sagebrush ecosystems is more dependent on the initial introduction of seed than on disturbance. Some plants are expanding their range to new locations and, therefore, would represent the "introduction" phase of invasion (Groves 1986). These include yellow starthistle, squarrose knapweed (*Centaurea triumfettii*), Mediterranean sage (*Salvia aethiopis*), dyer's woad (*Isatis tinctoria*), and medusahead. They may dominate local sites in portions of states but continue to be discovered in new locations (Table 1).

The colonization phase (Groves 1986) is represented by species that are already found throughout the Intermountain West. These may be locally dense in certain regions of a state but only sparsely represented in other locations. Examples of species that fit this description include leafy spurge (*Euphorbia esula*), whitetop (*Cardaria pubescens*), and some species in the knapweed complex – diffuse knapweed (*Centaurea diffusa*) and spotted knapweed (*C. biebersteinii*). These species are expanding their populations where they currently exist while continuing to spread to new locations.

Within the sagebrush ecosystems, however, the annual exotic grasses exemplify the "naturalization" phase of invasions (Groves 1986) and are the most problematic for management within the region. Species within this group include cheatgrass, Japanese brome (*Bromus japonicus*), and soft brome. These species have attained their geographic distribution in the region. They are now commonly found mixed with native species, even in locations undisturbed by livestock (Svejcar and Tausch 1991, Kindschy 1994). These species are so prevalent throughout the region that it would be difficult or economically burdensome for people to control them; therefore, they are not listed as state or federal noxious weeds.

ANNUAL EXOTICS: MECHANISMS TO INVADE AND DOMINATE

A plant invader that achieves the three phases of the invasion process – introduction, colonization, and naturalization (Groves 1986) – will be widespread in the ecosystem. A combination of both the invader's traits and the ecosystem's conditions allows for species to successfully move through the phases of an invasion. However, generalized characteristics of either the species or its potential new environment that would help predict invasions have often led to more exceptions than general rules (Lodge 1993). Since cheatgrass and medusahead are prevalent throughout the sagebrush ecosystems (e.g., Pellant and Hall 1994), I will concentrate my discussion of mechanisms for invasion and dominance on the characteristics of these species, coupled with the characteristics of the ecosystem.

The original introductions of both species into the sagebrush ecosystems likely occurred in the late 1800s. The cheatgrass introduction was probably associated with the import of contaminated cereal grain seed, since the earliest collections were found around wheat-growing areas (Mack 1986). Less is known about the original introductions of medusahead. The earliest collections occurred near Roseburg, Oregon, between 1884 and 1887 (Furbish 1953, Turner et al. 1963). The first collection in the Intermountain West occurred near Steptoe Butte, Washington (St. John 1937). One could argue that contaminated cereal grain seed was to blame for the Washington introduction since this was a wheat-growing region, but others have speculated that the seed was introduced on the fur of imported animals (Hilken and Miller 1980).

Cheatgrass is currently more prevalent than medusahead in sagebrush ecosystems. Mack (1981) estimated the complete range of cheatgrass in the Intermountain West at 40 million ha (99 million acres). This estimate is probably conservative, since Pellant and Hall (1994) surveyed BLM lands and estimated that one million ha (2.5 million acres) of these lands in Idaho, Nevada, Oregon, Utah, and Washington are dominated by cheatgrass (>60% of the species composition by weight). They estimated that 31.8 million ha (78.5 million acres) of BLM lands (about 80% of these lands in the 5 states) have the potential for cheatgrass to become dominant.

Although medusahead was introduced at about the same time as cheatgrass, it has spread more slowly than cheatgrass. Miller et al. (1999) estimated that medusahead occurs on 400,000 ha (988,000 acres) throughout its complete range; however, much of that land is in California. Pellant and Hall's (1994) survey of BLM lands in the Intermountain West estimates that medusahead occupies approximately 167,000 ha (412,500 acres) in Idaho and Oregon. This species has not reached its potential distribution in the region since new introductions have been reported in several locations in Utah and Nevada (Horton 1991, Young 1992).

The difference between cheatgrass and medusahead in rate of spread may relate to their genetics. Both species have self-mating reproductive systems. We know very little about the genetics of medusahead, but we know that several genetic strains of cheatgrass from different regions of Eurasia have been introduced into the Intermountain West (Novak et al. 1991, Novak and Mack 1993, Pyke and Novak 1994). These multiple cheatgrass introductions may provide greater adaptations to establish and survive in a wider range of environments. Future research might investigate whether medusahead's slow expansion relates to less genetic diversity in the form of fewer introductions from its native environment.

INVASION MECHANISMS

Both cheatgrass and medusahead are obligate annual grasses (only rare exceptions have been noted [Harris 1967]); therefore, population sustainability, as well as population initiation, requires available seeds. The invasion process of an annual plant requires the combination of seed arrival to the site (dispersal dynamics) plus germination and survival of the plant until successful reproduction (Cousens and Mortimer 1995). To understand how these species invade the sagebrush ecosystem, we must understand their seed production and dispersal.

Both cheatgrass and medusahead are highly plastic in their production of seeds. Under a wide range of environmental conditions (Rice and Mack 1991), most individuals will produce at least one seed. To exemplify this point, one-month-old plants can withstand weekly severe grazing (defoliation to the soil surface) and still produce viable seeds if given time to reproduce (eight weeks) at the end of the growing season (Pyke 1987). Both species have indeterminate reproduction on their inflorescences, an advantage in variable environments (Pyke 1986) that allows for a wide range in seeds per tiller. The number of tillers produced per plant also contributes to the total seed production per individual and tends to be regulated by the density of the neighboring plants. Tiller production generally varies between 1 and 25 tillers per individual for dense vs. sparse neighborhoods (Hulbert 1955, Miller 1996). In a fertile yet sparse system, Sharp et al. (1957) reported a medusahead plant produced 133 tillers.

The neighborhood of species that grow with these annuals also influences their reproduction. Reichenberger and Pyke (1990) showed that reproduction of cheatgrass declined, depending on the species of the neighbor (sagebrush [3.6 cheatgrass seeds/plant] < bluebunch wheatgrass [*Pseudoroegneria spicata*] [6.1 seeds/plant] < desert crested wheatgrass [*Agropyron desertorum*] [8.1 seeds/plant]).

The soils on which plants grow may contribute to their success. Young (cited in Miller et al. 1999) speculates that medusahead has not been found in many Nevada locations because the salt desert communities in the valleys and the coniferous forests in the mountains act as barriers for establishment. Miller (1996) found medusahead reproduction at a site with clay soils was higher than medusahead on loam soil; however, climate may have also contributed to this result.

The variable yet temporally constant seed production of these annual grasses provides the necessary base resource for invasion to occur. The dispersal of seed to new locations is the next component that contributes to invasion success. Both species have similar mechanisms for long- and short-distance dispersal. I previously mentioned the speculation that both animal transport and crop seed contamination are likely avenues for the original introductions. These mechanisms are potential sources for continued spread.

Although regulations have curtailed the spread of many invasive plants with crop seeds, care must be taken to ensure the purity of seeds used in revegetation, restoration, or rehabilitation projects. Since many people desire native seeds on such projects, seeds are often collected in the wild, making the project vulnerable to contamination by nontarget seeds. Care should be taken to request that seed not contain invasive weeds like cheatgrass (Table 1) and to have a professional seed lab check purity prior to seeding.

Animal transport of invasive-plant seeds has been documented in many species. Although animal transport is likely in cheatgrass and medusahead, clear documentation of it has not been reported. The barbed awns of both species make them suitable for transport on animal fur. Many review papers have speculated about sheep or livestock in general as the dispersal vectors for these species (Mack 1981, Yensen 1981, Young 1992), but no studies have attempted to quantify this.

Recreational activities may result in seed dispersal. Seed transport on clothing is a common occurrence in sites with mature cheatgrass. The seeds become lodged in clothing, such as socks or shoes, and are moved along with the people. It is also common for seeds to become lodged in the chassis' of automobiles and all-terrain vehicles. In all cases, seeds may not dislodge until they have moved hundreds of miles.

The mechanisms for short-distance dispersal in both species involve secondary dispersal by wind once the seeds drop from the plant. Single spikelets of cheatgrass or whole inflorescences of medusahead are often blown across the soil until they hit an obstruction (e.g., litter or soil crack) (Turner et al. 1963, Bookman 1983).

The last component of an introduction is the plant's ability to germinate, emerge, and survive in the environment in which it is now found. The ability for cheatgrass to emerge in almost any season, provided there is adequate moisture (Mack and Pyke 1984), and to maintain high survival and reproduction even under intense and frequent herbivory (Pyke 1986, 1987) provides this species with an excellent mechanism to invade. Medusahead appears more restricted by soil texture and precipitation. It is more successful on clay than on loam soils (Young 1992, Miller 1996). It also seems to require more moisture than cheatgrass to successfully reproduce (Miller et al. 1999). These requirements tend to restrict it to clay soils with >30 cm of annual precipitation.

Dominance Mechanisms

The mechanisms that provide an introduction advantage to cheatgrass and medusahead in sagebrush ecosystems are generally the same types of mechanisms that confer an introduction advantage to other species. However, the combinations of mechanisms that allow these species to dominate the sagebrush ecosystems are more specific to these species, or at least to annual grasses in general, than the introduction mechanisms. Cheatgrass and medusahead become dominant in this ecosystem because of three general mechanisms. First, they are capable of occupying spatial or temporal niches that other vascular species commonly do not occupy. Second, they are capable of tolerating or avoiding disturbances that negatively impact many native plants in the ecosystem. Third, they compete successfully for resources with other vascular plant species in the ecosystem.

Spatially, these annual grasses occupy and expand to fill the interspaces between vascular plants. Nonvascular plants that make up the biological soil crust, such as mosses, lichens, algae, and cyanobacteria, historically occupied these interspaces. Biological soil crusts are easily damaged by trampling, especially when they are dry (Harper and Marble 1988, West 1990). Although one study has indicated that biological soil crusts may reduce cheatgrass establishment (Larsen 1995), others have shown that these crusts can enhance establishment, even with exotic plants (Harper and Marble 1988). Further research is needed to investigate the role of these crusts in vascular plant establishment.

Annual grasses are capable of establishing in a wide range of spatial locations. Cheatgrass seeds and seedlings appear to exist and grow under shrub canopies and in the interspaces (Young and Evans 1975). Soil cracks and seed burials to depths up to 2.0 cm are equally safe sites for emergence (Bookman 1983). Litter, especially from itself (Young et al. 1971), enhances medusahead germination.

Temporally, these annual grasses are capable of germinating in either the autumn, winter, or spring (Young 1992). Root growth of both annuals is quicker than bluebunch wheatgrass, a common dominant native grass (Harris and Wilson 1970, Harris 1977). This faster growth allows the annuals to establish their root systems before the natives. Bookman and Mack (1982) have shown that cheatgrass is capable of adjusting the placement of its roots, depending on the root placement of its neighbor. This plasticity in root placement may confer an advantage to cheatgrass when initially establishing in a community. Both annual grasses are able to capture nutrient pulses as they occur in the sagebrush ecosystem. Phenologic differences between these species allow cheatgrass to benefit more from early season pulses, while medusahead benefits from late season pulses (Bilbrough and Caldwell 1997).

The dominance of invasive exotic annual grasses in sagebrush ecosystems is related to the annuals' efficient mechanisms for withstanding disturbances to the ecosystem by livestock grazing and fire. Medusahead and cheatgrass are thought by many to use different mechanisms (avoidance for medusahead and tolerance for cheatgrass) when confronted with livestock grazing; however, research results are mixed regarding grazing avoidance yet clearly support the grazing tolerance hypothesis (as defined by Briske 1991).

Nutritionally, both cheatgrass and medusahead provide similar moisture content, crude protein, crude fat and fiber, and lignin content to that of desirable grasses, but medusahead is high in silica. Based upon this high silica content, Bovey and others (1961) concluded that medusahead was unpalatable to livestock at all growth stages. However, the only grazing experiment indicated that sheep used the plant when it was green (Lusk et al. 1961). Other studies have calculated the impact on livestock grazing using the compositional proportion of medusahead in the plant community, but actual grazing evaluations that show non-use or preferences do not appear in the literature (Higgins and Torell 1960, Torell et al. 1961). If medusahead's silica content provides an avoidance mechanism, this may occur when litter becomes deep, causing livestock to avoid the plant, as Hironaka speculated (cited in Hilken and Miller 1980).

Regardless of the potential for medusahead to avoid grazing, cheatgrass clearly is tolerant of grazing (Pyke 1986, 1987). Cheatgrass exhibits typical morphological characteristics of grazing-tolerant plants that allow them to regrow following defoliation (Archer and Pyke 1991). Using livestock to control cheatgrass has been reported in one review, but it notes that grazing must be continued until plants reach the purple stage and must be repeated for several years (Mosley 1996). However, Daubenmire (1940) noted that if grazing is not continued, cheatgrass would quickly return.

Fire is a natural disturbance in the sagebrush ecosystem, but the introduction of exotic annual grasses has shortened fire cycles and led to a reduction in the shrub component (D'Antonio and Vitousek 1992). Before the introduction of exotic annual grasses, the natural fire-return intervals were thought to be between 20 and 100 years, depending on the local climate and subspecies of sagebrush (Burkhardt and Tisdale 1976, Wright and Bailey 1982). Now, with annuals like cheatgrass in the ecosystem, the return interval has shortened to as few as five years under some conditions (Whisenant 1990). Similar to cheatgrass, medusahead provides fine fuels for wildfires, and the annual life span of both species results in large amounts of dry litter available to burn during the late summer (Young et al. 1971, Whisenant 1990). For medusahead, prescribed fire has been successfully used as a temporary control measure before revegetation (Miller et al. 1999). The ultimate result of frequent fires in sagebrush ecosystems is the elimination of fire-sensitive shrubs such as sagebrush (West 1983*a*,*b*).

The last mechanism of cheatgrass and medusahead that contributes to their eventual dominance in sagebrush ecosystems is their ability to compete successfully with native plants for available nutrients and water. The early germination and rapid root growth of both species are thought to contribute to this ability (Harris 1967, Harris and Goebel 1976, Harris 1977). The competitive advantage of these exotic annual grasses is most apparent when they compete with seedlings, even with competitive introduced forage grasses (Aguirre and Johnson 1991, Francis and Pyke 1996).

Since cheatgrass and medusahead overlap in their distribution, they do compete for resources in those locations. Controlled experiments have yielded mixed results regarding the competitive outcomes of mixed populations of medusahead and cheatgrass. In low-nutrient environments, either medusahead excelled or the two species were similar; but under high nitrogen levels, cheatgrass was most successful (Dakheel et al. 1993). Climate and soils may control the success of one species over the other where they coexist. Medusahead tends to dominate clay soils with more than 30 cm precipitation, whereas cheatgrass dominates coarser-textured soils in drier climates (Dakheel et al. 1993, Miller et al. 1999).

CONCLUSIONS

There is no guarantee that the exotic plants of today are the only species with which managers of sagebrush ecosystems must contend. Preventing further spread of exotic plants will require a concerted effort on the part of land managers and land users alike. They must consider the various modes of introductions for these species and use precautionary measures when moving throughout the region. Despite their best efforts, invasive exotic plants may continue to spread. Therefore, educating the public on the identification of exotic plants and on the plants' modes of introduction, and then applying this knowledge when people use the land, should help to slow the spread of exotic plants and to retain the native ecosystems for future generations to use and enjoy.

Table 1. List of the common weeds, noxious weeds, or invasive exotic plants in sagebrush ecosystems of the Intermountain West. The growth form (herb, shrub, or grass), life history strategy (annual, biennial, or perennial), the origin of the plant, states within the Intermountain West where it is listed as a noxious weed, and comments on the geographic distribution within the region are given for each species (Whitson et al. 1996, Rice 1997, Sheley and Petroff 1999, USD, NRCS 1999). Those species listed in bold are highly invasive and competitive in sagebrush ecosystems.

Family	Species	Common Name	Growth Form	Life History	Origin	Noxious	Distribution
Asteraceae	***Acroptilon repens* (L.) DC. [formerly *Centaurea repens* L.]**	**Russian knapweed**	**herb**	**perennial**	**Eurasia**	**CA, ID, MT, NV, OR, UT, WA, WY**	**Locally dense throughout Northwest**
Asteraceae	*Arctium minus* Bernh.	common burdock, lesser burdock	herb	biennial, perennial	Europe	NV, WY	Widespread
Asteraceae	*Carduus acanthoides* L.	spiny plumeless thistle	herb	biennial, perennial	Eurasia	CA, WA, WY	Locally dense in ID, MT, WA, & WY
Asteraceae	*Carduus nutans* L.	musk thistle, nodding plumeless thistle	herb	biennial, perennial	Eurasia	CA, ID, NV, OR, UT, WA, WY	Widespread
Asteraceae	*Carthamus lanatus* L.	distaff thistle	herb	annual	S. Europe, N. Africa	CA, OR	Occasionally in CA, OR, & NV
Asteraceae	***Centaurea biebersteinii* DC. [formerly *Centaurea maculosa* Lam.]**	**spotted knapweed**	**herb**	**biennial, perennial**	**Eurasia**	**CA, ID, MT, NV, OR, UT, WA, WY**	**Widespread**
Asteraceae	***Centaurea diffusa* Lam.**	**diffuse knapweed**	**herb**	**annual, perennial**	**Eurasia**	**CA, ID, MT, NV, OR, UT, WA, WY**	**Widespread**
Asteraceae	*Centaurea iberica* Trev. ex Spreng.	Iberian starthistle	herb	annual, biennial	Europe	CA, NV, OR	Locally dense in WA, OR, CA, & WY
Asteraceae	***Centaurea solstitialis* L.**	**yellow starthistle**	**herb**	**annual**	**Europe**	**CA, ID, MT, NV, OR, UT, WA**	**Locally dense in OR, WA, ID, CA & NV**
Asteraceae	***Centaurea triumfettii* All. [formerly *C. virgata* Lam.]**	**squarrose knapweed**	**herb**	**perennial**	**Europe**	**CA, OR, UT**	**Locally dense in UT, OR, CA**
Asteraceae	***Chondrilla juncea* L.**	**rush skeletonweed, hogbite**	**herb**	**perennial**	**Eurasia**	**CA, ID, MT, OR, WA**	**Locally dense throughout Intermountain West**
Asteraceae	***Cirsium arvense* (L.) Scop.**	**Canadian thistle**	**herb**	**perennial**	**Eurasia**	**CA, ID, MT, NV, OR, UT, WA**	**Widespread**
Asteraceae	***Crupina vulgaris* Cass.**	**common crupina**	**herb**	**annual**	**Eurasia, N. Africa**	**CA, ID, MT, OR, WA, USA**	**Locally dense in CA, ID, OR, WA**

Table 1 (Cont.)

Family	Species	Common Name	Growth Form	Life History	Origin	Noxious	Distribution
Brassicaceae	*Brassica nigra* (L.) W.D.J. Koch	black mustard	herb	annual	Europe		Widespread
Brassicaceae	*Brassica rapa* L.	rape mustard	herb	annual, biennial	Europe		Widespread
Brassicaceae	*Capsella bursa-pastoris* (L.) Medik.	shepherd's purse	herb	annual	Europe		Widespread
Brassicaceae	***Cardaria pubescens* (C.A. Mey.) Jarmolenko**	**whitetop**	**herb**	**perennial**	**Eurasia**	**CA, OR, UT, WA, WY**	**Widespread**
Brassicaceae	*Descurainia sophia* (L.) Webb ex Prantl	flixweed, herb sophia	herb	annual, biennial	Europe		Widespread
Brassicaceae	***Isatis tinctoria* L.**	**dyer's woad**	**herb**	**biennial, perennial**	**Europe**	**CA, ID, MT, NV, OR, UT, WA, WY**	**Locally dense in ID, MT, UT, WY, NV, CA**
Brassicaceae	***Lepidium latifolium* L.**	**perennial pepperweed, tall whitetop**	**herb**	**perennial**	**Eurasia**	**CA, ID, NV, OR, UT, WA, WY**	**Locally dense, transitions from meadow to upland**
Brassicaceae	*Sisymbrium altissimum* L.	tumble mustard	herb	annual, biennial	Europe		Widespread
Brassicaceae	*Thlaspi arvense* L.	field pennycress	herb	annual	Europe		Widespread
Caryophyllaceae	*Gypsophila paniculata* L.	baby's breath	herb	perennial	Europe	CA, WA	Locally dense in ID, MT, WA, & OR
Caryophyllaceae	*Vaccaria hispanica* (P. Mill.) Rauschert	cowcockle, cow soapwort	herb	annual	Europe		Widespread
Chenopodiaceae	*Chenopodium album* L.	lambsquarters	herb	annual	Eurasia		Widespread
Chenopodiaceae	*Halogeton glomeratus* (Bieb.) C.A. Mey.	halogeton	herb	annual	Asia	CA, NV, OR	Widespread, alkaline soils
Chenopodiaceae	*Salsola kali* L. ssp. *tragus* (L.) Celak.	Russian thistle	herb	annual	Eurasia		Widespread
Clusiaceae	*Hypericum perforatum* L.	common St. John's wort, klamathweed	herb	perennial	Europe	CA, MT, NV, OR, WA	Widespread
Dipsacaceae	*Dipsacus fullonum* L.	Fuller's teasel, common teasel	herb	biennial, perennial	Europe		Widespread

Table 1 (Cont.)

Family	Species	Common Name	Growth Form	Life History	Origin	Noxious	Distribution
Euphorbiaceae	***Euphorbia esula* L.**	**leafy spurge, wolf's milk**	**herb**	**perennial**	**Eurasia**	**CA, ID, MT, NV, OR, UT, WA, WY**	**Widespread**
Geraniaceae	*Erodium cicutarium* (L.) L'Her. ex Ait.	redstem filaree, redstem stork's bill	herb	annual, biennial	Eurasia		Widespread
Lamiaceae	*Marrubium vulgare* L.	horehound	herb, subshrub	perennial	Europe		Widespread
Lamiaceae	***Salvia aethiopis* L.**	**mediterranean sage**	**herb**	**biennial**	**S. Europe, N. Africa**	**CA, NV, OR, WA**	**Locally dense in OR, also found in WA, ID, CA**
Poaceae	***Aegilops cylindrica* Host**	**jointed goatgrass**	**grass**	**annual**	**Europe**	**CA, ID, OR, WA**	**Locally dense in MT, eastern WA often associated with winter wheat**
Poaceae	***Bromus japonicus* Thunb.**	**Japanese brome**	**grass**	**annual**	**Europe**		**Widespread**
Poaceae	***Bromus mollis* L.**	**soft brome**	**grass**	**annual**	**Europe**		**Widespread**
Poaceae	***Bromus rubens* L.**	**red brome, foxtail brome**	**grass**	**annual**	**Europe**		**Locally dense in CA, ID, NV, OR, UT, WA**
Poaceae	*Bromus secalinus* L.	cheat, rye brome	grass	annual	Europe		Widespread
Poaceae	***Bromus tectorum* L.**	**cheatgrass, downy brome**	**grass**	**annual**	**Eurasia**		**Widespread**
Poaceae	*Poa bulbosa* L.	bulbous bluegrass	grass	perennial	Europe		Widespread
Poaceae	***Taeniatherum caput-medusae* (L.) Nevski**	**medusahead**	**grass**	**annual**	**Eurasia**	**OR**	**Locally dense in CA, ID, NV, OR, and WA**
Poaceae	*Ventenata dubia* (Leers) Coss. & Durieu	ventenata	grass	annual	Eurasia		Locally dense in N. ID, E. WA and N.E. OR
Ranunculaceae	*Ranuculus testiculatus* Crantz	bur buttercup	herb	annual	Eurasia		Widespread
Scrophulariaceae	***Linaria dalmatica ssp. dalmatica* (L.) P. Mill.**	**Dalmatian toadflax**	**herb**	**perennial**	**SE Europe**	**CA, ID, MT, OR, WA, WY**	**Locally dense throughout Northwest**
Scrophulariaceae	*Linaria vulgaris* P. Mill.	butter-and-eggs, yellow toadflax	herb	perennial	Eurasia	ID, OR, WA, WY	Scattered throughout Northwest
Scrophulariaceae	*Verbascum thapsus* L.	common mullein	herb	biennial, perennial	Europe	WA	Widespread

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STRUCTURE AND FUNCTION OF BIOLOGICAL SOIL CRUSTS

Jayne Belnap

INTRODUCTION

In arid and semiarid lands throughout the world, the cover of vegetation is generally sparse or absent. Open spaces between the higher plants are not bare of autotrophic life but usually covered by a community of highly specialized organisms. This soil surface floral community consists of cyanobacteria, green algae, lichens, mosses, microfungi, and other bacteria. Cyanobacterial and microfungal filaments weave throughout the top few millimeters of soil, gluing loose soil particles together to form a biological crust. These crusts occur in all hot, cool, and cold arid and semiarid regions. They may constitute up to 70% of the living cover (Belnap 1994) and have only recently been recognized as having a major influence on terrestrial ecosystems. These communities are also referred to as cryptogamic, cryptobiotic, microbiotic, or microphytic soil crusts (Harper and Marble 1988).

Physical soil crusts are also a major structural feature in many arid regions and are often confused with biological soil crusts. Most physical crusts are formed by raindrops hitting unprotected soil surfaces, which breaks apart surface aggregates. Smaller particles then wash into spaces between larger particles, thus clogging soil pores and reducing infiltration rates by as much as 90%. In general, rain-formed crusts are less than 5 mm thick. This layer is often harder than the rest of the soil because it is drier and compounds such as salts, lime, and silica are often deposited at the surface as water evaporates. With large pores absent, these crusts increase water runoff and soil erosion and prevent the emergence of seedlings. Thus, physical crusts play a very different role in arid ecosystems than do biological crusts (Lemos and Lutz 1957).

MICROSTRUCTURE

Lichens and mosses are easily seen without aid of magnification. However, much of the structure and function of crusts depends on cyanobacteria, green algae, and microfungi, which are often too small to be seen without a microscope. In most desert soils, cyanobacteria contribute the most to crust microstructure.

Cyanobacterial filaments confer structural integrity to the soils in which they occur. When wetted, the sheath of filamentous cyanobacteria swell, expelling the living filaments and leaving behind empty sheath material. These filaments often string sand and clay particles together, much like fibers in fiberglass. Depending on environmental conditions and soil textures, cyanobacterial sheaths may be found at depths of 10 cm below the soil surface (Belnap and Gardner 1993). As aeolian and water-borne materials are trapped in the polysaccharide sheaths of cyanobacteria on the soil surface, old sheaths are gradually buried. Thus, influence on water-holding capacity and soil stability may extend far below the depth to which light can penetrate, unless sheaths are crushed. If sheath-soil connections are broken by trampling or vehicles, these sheaths are no longer living and therefore cannot be repaired.

ECOLOGICAL ROLES – CARBON AND NITROGEN FIXATION

Biological soil crusts are an important source of fixed carbon on sparsely vegetated areas throughout the West (Beymer and Klopatek 1991). While vascular plants provide organic matter to soils directly underneath them, large interspaces between plants have little opportunity to receive such input. Carbon contributed by soil crusts helps keep plant interspaces fertile and thus provides energy sources for other microbial populations.

The dominant components of biological soil crusts are photosynthetic organisms that require sunlight. When soils are dry, the bulk of the cyanobacterial biomass is at 0.2 - 0.5 mm, with bundles found down to 4 mm where sufficient light for net carbon gain is available but UV exposure is reduced (Garcia-Pichel and Belnap 1996). Carbon fixation rates are dependent on moisture and temperature (Rychert et al. 1978; Nash et al. 1982*a*,*b*; Lange et al. 1997). Most crustal species increase photosynthetic rates with increasing temperatures up to about 26-28°C, after which rates decline.

Nitrogen concentrations are known to be low in desert ecosystems relative to other ecosystems. Total atmospheric input of nitrogen over the past 10,000 years has been conservatively estimated at about 3 kg/m^2 (ignoring cyanobacteria inputs), with 77% lost through wind erosion, ammonia volatilization, nitrification, and denitrification (Peterjohn and Schlesinger 1990).

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Extensive surveys in cold deserts have revealed only a few nitrogen-fixing plants (Farnsworth et al. 1976). As nitrogen can limit net primary productivity in many desert ecosystems, normal nitrogen cycles are critical to the fertility of semiarid soils and in preventing desertification (Dregne 1983).

Cyanobacteria and cyanobacterial-containing soil lichens can be an important source of both fixed nitrogen for plants and soils in desert ecosystems (Evans and Ehleringer 1993, Belnap 1995). Nitrogen inputs from biological soil crusts have been estimated from 1 to 100 kg/ha annually (Harper and Marble 1988), with the lowest estimates still almost 10 times atmospheric input estimates. Nitrogen fixation is highly dependent on past and present water and light regimes, as well as species composition (Rychert et al. 1978, Belnap 1996), with maximum fixation at approximately 26°C and above 20% moisture. Past disturbance history is also a critical factor in determining fixation rates (Belnap 1995, 1996).

Five to 88% of N fixed by *Nostoc* has been shown to leak into the surrounding substrate (Magee and Burris 1954, Belnap et al. 1997). Nitrogen leaked from these organisms is available to nearby vascular plants (Mayland and MacIntosh 1966). Vascular plants growing in biologically crusted areas show higher tissue concentrations of nitrogen when compared to plants in uncrusted soils (Harper and Pendleton 1993; Belnap 1994, 1995; Belnap & Harper 1995). As with carbon, crusts contribute nitrogen to soils both under plants and in plant interspaces, thereby counteracting the tendency of nutrients to concentrate around perennial plants.

EFFECTS ON VASCULAR PLANTS

Germination and establishment: The presence of crusts can affect vascular plant germination and establishment. While small cracks and crannies on the soil surface are sufficient for small-seeded plants to lodge and germinate, most large-seeded plants need some cover by soil or vascular plant litter. In deserts where plant litter is often limiting in interspaces, large native seeds generally have self-burial mechanisms (such as hygroscopic awns) or are rodent-cached. Plants adapted to loose, moving soils (such as sand dunes) or deep litter (forests) accomplish this passively. However, exotic species may lack such adaptations. As crusts stabilize soils, germination can be inhibited in sites with well-developed crusts and low plant litter, as was recently demonstrated for the annual exotic cheatgrass (*Bromus tectorum*), in both the field and the laboratory (Kaltenecker 1997). Once the seeds germinate, biological soil crusts show no barrier to seedling root penetration (J. Belnap, USGS Biological Resources Division, and R.L. Pendleton and S.E. Meyer, USDA Forest Service Shrub Sciences Laboratory, unpublished data). Seedling germination *per se* has not been shown to limit species density or presence in desert plant communities. Rather, many studies worldwide suggest that vascular plant cover is most often controlled by water and/or nutrient availability rather than other site factors (Mabbutt and Fanning 1987, Tongway and Ludwig 1990, Dunkerley and Brown 1995).

A recent review of the literature regarding the survival and biomass of plants in crusted soils compared to uncrusted soils shows that all perennial plants in cool deserts are either enhanced or not affected by the presence of biological soil crusts (Belnap et al. 2000). This included both fine- and coarse-textured soils. No study showed a negative relationship between crusts and vascular plant growth. Numerous other authors have reported similar findings (reviewed in Harper and Marble 1988). On the other hand, the presence of perennial plants may aid the survival of crustal components by increasing surface moisture due to shading.

Nutrient levels of plants growing on crusted soil generally show higher concentrations and/or greater total accumulation of various essential nutrients when compared to plants growing in adjacent, uncrusted soils. In southeast Utah, leaf tissue nitrogen in annual, biennial, and perennial species was 9 to 31% higher in crusted areas. Dry weights were greater as well (Belnap 1995, Belnap and Harper 1995). This was verified with greenhouse experiments (Harper and Pendleton 1993). Other authors have obtained similar results with other species (Shields and Durrell 1964, Brotherson and Rushforth 1983).

Several mechanisms have been postulated to explain this effect. Crusts accumulate nutrient-rich fine soil and organic matter (Fryberger et al. 1988, Verrecchia et al. 1995). Cyanobacterial sheath material is often coated with negatively charged clay particles. Positively charged macro-nutrients bind to these particles and are thus prevented from leaching from the soil profile (Belnap and Gardner 1993). These clay particles are more nutrient-rich than sand (Black 1968). Compounds in the gelatinous sheath material of cyanobacteria are able to chelate elements essential for their growth, e.g., iron, copper, molybdenum, zinc, cobalt, and manganese (Lange 1974). It is also possible that nutrient differences are a result of a thermal effect, as dark crusts would be warmer than lighter uncrusted soils and uptake of nutrients would occur at a higher rate. Herbivores and other consumers may benefit directly from the enhanced nutrient status of these ecosystems (Belnap and Harper 1995). Indirect effects include positive correlations between soil mycorrhizae and microarthropod populations with the presence of well-developed biological soil crusts (Harper and Pendleton 1993).

WATER RELATIONS

The effect of biological soil crusts on soil water relations is highly variable between different regions, soils, and climatic regimes. Crustal development (e.g., cyanobacterial, lichen, moss), climatic regimes, the amount of surface roughness, time since destructive disturbance, soil texture, and soil structure can all heavily influence hydrologic cycles at a given site. Soil texture is especially important and can override any effect of biological soil crusts. For instance, soils with high shrink-swell clays have low infiltration rates and sandy soils have high infiltration rates, regardless of the biological soil crusts present.

Results of research conducted under a variety of soil and climate conditions around the world show the variable and interactive effects of biological soil crusts and soil properties. While the presence of the mucilaginous cyanobacteria can decrease soil permeability, increased surface roughness can increase water pooling and residence time. Consequently, in cool and cold deserts, where frost-heaving is common and biological soil crusts greatly increase soil-surface roughness, the presence of biological soil crusts generally increases the amount and depth of rainfall infiltration. In warm deserts, where frost-heaving is not present and biological soil crusts are relatively flat, the influence of crusts on infiltration rates is dependent mostly on soil type, with crusted sandy soils showing a greater relative reduction (though absolute rates are still higher) than crusted fine-textured soils (Warren 2000).

Though overall infiltration of precipitation is critical for plant growth, where the water infiltrates can also be critical in maintaining plant community structure. Recent work done on banded vegetation has shown that water infiltration and runoff patterns can be important in maintaining vegetative community structure in hyper-arid zones. Biological soil crusts cover inter-band soils. When these inter-band biological soil crusts are disrupted, water infiltration increases between vegetated areas. This results in less water reaching the vegetated bands, causing large die-offs. This was also seen in Israel, where vegetation died when water infiltration was increased in plant interspaces (E. Zaady, Ben-Gurion University of the Negev, personal communication).

The effect of biological soil crusts on soil moisture is also variable. Soils under biological soil crusts showed deeper water penetration into the profile and greater availability during drought (Brotherson and Rushforth 1983, Abrahams et al. 1988). The ability of the crust to seal the soil surface and reduce evaporation due to high clay and silt concentrations in the crusts has been repeatedly proposed (Danin 1978, Brotherson and Rushforth 1983, Williams et al. 1995*a*) and recently supported by research specifically designed to address the issue (Verrecchia et al. 1995). However, this can vary. In Utah and Mexico, soil moisture was less under disturbed crusts than intact crusts (Harper and Marble 1988, Meyer and Garcia-Moya 1989). Increased soil temperature, through the absorption of solar energy by black crusts, may increase soil moisture evaporation rates (Harper and Marble 1988).

SOIL STABILIZATION

Wind and water can be major erosive forces in deserts, as sparse vegetation leaves large soil spaces unprotected by plant litter or vegetative cover (Goudie 1978). Interspace soils in deserts are most often stabilized by rocks or biological soil crusts. Polysaccharides extruded by the cyanobacteria and green algae, in combination with lichen and moss rhizines, entrap and bind soil particles together, increasing the size of soil aggregates. As soil aggregates get larger, they are heavier, have a greater surface area, and are therefore more difficult for wind or water to move. The presence of biological soil crusts enables otherwise loose, sandy soils to stay in place on steep slopes and stabilizes pockets of very shallow soil (reviewed in Harper and Marble 1988, Belnap and Lange 2000). Globally, many authors have reported that the presence of biological soil crusts reduces soil susceptibility to water erosion through reduced raindrop erosion and sediment loss from sites (Foth 1978, Harper and Marble 1988, Alexander and Calvo 1990, Eldridge 1993, Eldridge and Greene 1994, Ladyman and Muldavin 1994). Biological soil crusts are unambiguously effective in reducing wind erosion of soil. All studies have shown that crust cover reduces wind erosion by requiring much higher wind speeds to initiate soil particle movement (Williams et al. 1995*b*; McKenna-Neuman et al. 1996; Belnap and Gillette 1997, 1998). Resistance to water and wind erosion parallels biological crust development. The degree to which different types of crusts protect the soil surface from both wind and water erosion is: bare soil < algal crust < lichen/moss crust (Tchoupopnou 1989; Kinnell et al. 1990; Eldridge and Greene 1994; Belnap and Gillette 1997, 1998).

EFFECTS OF DISTURBANCE

Many uses of deserts result in impacts to biological soil crusts. The greatest impacts come from off-road vehicles, both military and civilian; trampling by livestock and people; and various mining activities. Effects of these activities are especially noticeable at sites with highly erodible soils with high topographic relief. Surface disturbance generally results in changes in species composition of soil crusts. While multiple species of soil lichens and mosses, as well as 4 or more species of cyanobacteria, can be found in untrampled areas on most soil types, no lichens and only 1 species of cyanobacteria are generally found in directly adjacent trampled areas (Belnap 1995).

Trampled surfaces are generally flat. Flattened surfaces do not slow water or wind velocity, nor does sediment have an opportunity to settle out; thus, more sediment is lost from trampled sites than untrampled sites. Water residence time on smooth surfaces is shorter and water infiltration reduced (Harper and Marble 1988). Trampling breaks cyanobacterial connections, compromising soil stability. Arid soils with little tendency to form inorganic aggregates (e.g., sandy soils) are more susceptible to stresses when dry, while soils with inorganic crusting are more susceptible to impacts when soils are wet. Soil formation is extremely slow in deserts, taking 5,000 to 10,000 years (Dregne 1983). Compressional disturbances to the crusts greatly decrease resistance to wind erosion for all soil types, regardless of the disturbance regime or soil type, as cyanobacteria and lichens are brittle when dry and crush easily. Vehicle tracks result in greater damage than hoof prints on a given soil type. After 10 years of recovery, sandy soils tested in southeast Utah were still susceptible to wind erosion at commonly occurring wind speeds, while fine-textured soils in southern New Mexico showed much quicker recovery (Belnap and Gillette 1997, 1998; Herrick, USDA Agricultural Research Service, and Belnap, USGS Biological Resources Division, unpublished data). Nearby biological soil crusts can also be buried by blowing sediment, resulting in the death of the photosynthetic organisms (Belnap 1995, 1996). Because over 75% of the photosynthetic biomass and almost all photosynthetic productivity are from organisms in the top 3 mm of these soils, very small soil losses can reduce site fertility and soil stability.

Nutrient Cycles: Crust disturbance can result in large decreases in soil nitrogen through a combination of reduced input (Belnap et al. 1994; Belnap 1995, 1996; Evans and Belnap 1999) and elevated losses (Peterjohn and Schlesinger 1990). Reductions in input can continue long after disturbance has been removed: current long-term studies demonstrate a 42% decrease in soil nitrogen and 34% decrease in plant tissue nitrogen 25 years following release from grazing. This has severe implications for ecosystems that are dependent on biological crusts for nitrogen, such as the Colorado Plateau (Evans and Ehleringer 1993, Evans and Belnap 1999). Reduced fertility of systems is one of the most problematic aspects of desertification (Dregne 1983).

Albedo: Albedo is also of concern in semiarid and arid systems. When trampled crusts were compared to untrampled crusts, there was up to a 50% increase in reflectance across the spectrum. This represents a change in the surface energy flux of approximately 40 watts/m^2. Soil temperatures are up to 14°C lower on the lighter, trampled surface (Belnap 1995). Altered soil temperatures affect rates of carbon and nitrogen fixation; microbial activity; plant germination, growth, and nutrient uptake; and soil water evaporation (Harper and Marble 1988, Bush and Van Auken 1991). Food and other resources are often partitioned among ants, arthropods, and small mammals on the basis of surface temperature-controlled foraging times (Doyen and Tschinkel 1974, Wallwork 1982, Crawford 1991). Many small desert animals are weak burrowers, and soil surface microclimates are of great importance to their survival (Larmuth 1978). Consequently, altering surface temperatures can affect nutrient availability and community structure for many desert organisms, thus increasing susceptibility to desertification.

Fire: High-intensity fire will burn biological crusts, resulting in reduction of visible cover, biomass, and species diversity (Callison et al. 1985, Greene et al. 1990, Johansen 1993). The extent of damage depends on the type of plant community in which the crust occurs, the distribution of fuel, and thus fire intensities (Johansen 1993). Exotic annual grasses, primarily *Bromus* spp., have invaded semiarid and arid landscapes throughout western North America, homogenizing fuel distribution and drastically altering fire regimes (Whisenant 1990). Increases in both fuel amount and continuity have resulted in large, continuous fires. Biological crusts are lost from the community if fire-return intervals are shorter than the period required for the crusts to recover (Greene et al. 1990, Whisenant 1990).

EXOTIC PLANTS

Introduced annuals such as cheatgrass and medusahead wildrye (*Taeniatherum asperum*) appear to impose long-term threats to biological soil crust communities. Surveys in these plant communities show that the rich perennial moss/lichen community has generally been replaced with annual mosses and cyanobacteria. The mechanism by which the presence of annual grasses negatively affects the biological soil crusts is not clear but could include a decrease in available soil surfaces (via increased cover of vascular plant and plant litter), increased soil disturbance by small rodents responding to an increase in seed availability, increased fire frequency, increased soil turnover by increased populations of soil fauna, and/or increased soil disturbance by plant surface roots (Kaltenecker 1997).

RECOVERY FROM DISTURBANCE

Natural Recovery Rates

Species Composition: Recovery rates of biological soil crusts depend on the type and extent of disturbance and the availability of nearby inoculation material, as well as the temperature and moisture regimes that follow disturbance events. Recovery time is faster when crustal material is not removed, as pieces of remaining organisms are available to reinoculate recovering surfaces. Therefore, although most damage is done with the initial impact, recovery will be faster if disturbances are not repeated. Timing of the disturbance is also important. Damage is less severe when crusts are wet. In addition, if damage occurs when rain is imminent, then crustal organisms have an opportunity to reattach themselves before being blown away or buried. However, if disturbances occur before a long dry period, reattachment is not possible and much crustal material may be lost or too deeply buried for recovery. Size of disturbance can be important, especially if crustal material has been lost from the disturbed site. As inoculant must come from adjoining areas, the size of the perimeter area relative to the internal surface area of the disturbance can heavily influence recovery rates (Belnap and Eldridge 2000). In addition, recovery is slower if soils in adjacent areas are destabilized. Sediments from these areas can either bury adjacent crusts, leading to their death, or provide material for "sandblasting" nearby surfaces, thus increasing erosion rates and slowing recovery (McKenna-Neumann et al. 1996).

Cyanobacteria or green algae recover first. *Microcoleus* is generally the first species to appear. Cyanobacteria are mobile and can often move up through disturbed sediments to reach needed light levels for photosynthesis, while slow-growing lichens and mosses are incapable of such movement. Instead, they require stable soil surfaces for growth, and colonization of these components generally takes place after surfaces have been stabilized by cyanobacteria. *Collema*, a nitrogen-fixing lichen, is generally the first lichen to appear.

The recovery process is more rapid in regions where soil surface moisture lasts for a longer period of time. Sites with fine-textured soils such as silt loams retain surface soil moisture for a longer period than do coarse-textured, sandy, or gravelly soils. Depending on all the above-mentioned factors, estimates of recovery time in cool deserts ranges from 14 to 35 years for cyanobacterial biomass, 45 to 85 years for lichen cover, and 20 to 250 years for moss cover (Belnap and Eldridge 2000).

Enhanced Recovery Rates

The use of inoculants to speed up recovery of crusts works well (Lewin 1977, Tidemann et al. 1980, Ashley and Rushforth 1984, St. Clair et al. 1986). In an experiment reported from southeast Utah, all measured responses were significantly enhanced by inoculation (Belnap 1993, 1995, 1996).

EVOLUTIONARY HISTORY OF DISTURBANCE

Soil and plant characteristics of most Intermountain ecosystems suggest that they probably evolved with low levels of soil surface disturbance by ungulates. These characteristics include limited surface water; sparse vegetation; the presence of biological soil crusts, which are easily disrupted by trampling; and the dependence of these ecosystems on nitrogen provided by the biological soil crusts (Evans and Ehleringer 1993, Evans and Belnap 1999). Dung beetles, present globally in other systems with large ungulate populations, are lacking (Mack and Thompson 1982). Limited surface water would have kept ungulate populations small and generally limited to winter use of lower elevations, as is seen today (Parmenter and Van Devender 1995). Winter use results in lower impacts to biological crusts (Marble and Harper 1989), as soils are wet or soon to be wet. Bunchgrasses that lack adaptations to grazing such as tillering, secondary compounds, or high tissue silica content are dominant (Martin 1975, Stebbins 1981, Mack and Thompson 1982). In addition, shallow soils and low precipitation limit the distribution of burrowing vertebrate and invertebrate species. Thus, these systems may depend more heavily than other regions on soil surface integrity for natural ecosystem functioning. As a result, these deserts may be more negatively affected by soil surface disturbances than deserts that evolved with higher levels of surface disturbance.

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Restoration of Sagebrush Steppe Ecosystem Resource Values

NOXIOUS WEED CONTROL

Steven Dewey

ABSTRACT

Nonnative noxious weeds are advancing at an alarming rate on public and private lands. Current estimates of spread on federal lands alone range between 4,000 and 8,000 acres (1,620-3,240 ha) per day. Once established, noxious weeds crowd out native plants to form drastically altered plant communities of limited species diversity. Wildlife populations dependent on native plants are often adversely affected. Other negative ecological impacts can include increased soil erosion, water runoff, and wildfire frequency. Sagebrush steppe communities are among the ecosystems most vulnerable to invasion and degradation by invasive weeds. Weed species well suited to sagebrush steppe communities in the western United States include diffuse and squarrose knapweed (*Centaurea diffusa* and *C. triumfettii*), yellow starthistle (*C. solstitialis*), musk and Scotch thistle (*Cardus nutans*, *Onopordum acanthium*), Dalmatian and yellow toadflax (*Linaria dalmatica*, *L. vulgaris*), dyer's woad (*Isatis tinctoria*), hoary cress (*Cardaria draba*), leafy spurge (*Euphorbia esula*), rush skeletonweed (*Chondrilla juncea*), houndstongue (*Cynoglossum officinale*), black henbane (*Hyoscyamus niger*), and St. John's wort (*Hypericum perforatum*). Effective weed management requires protection of noninfested lands by preventing the introduction and establishment of new weed infestations and early detection and eradication of all new infestations before they spread. Management of larger infestations is accomplished through a strategy of containment and control, using an integrated and balanced combination of herbicides, cultural practices, and biological controls. Numerous herbicides are currently available for use on rangeland. Biocontrol agents exist for many invasive rangeland weed species, and more are being developed.

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CONTROLLING ANNUAL GRASSES WITH OUST® HERBICIDE

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INTRODUCTION

Introduced in the late 1800s, 2 annual grasses, cheatgrass (*Bromus tectorum* L.) and medusahead (*Taeniatherum caput-medusae* [L.] Nevski), spread rapidly through degraded shrub steppe communities (Stewart and Hull 1949, Hironaka 1963, Pellant and Hall 1994). Their competitive ability and the continuous mats of fine fuels produced by these 2 species contributed to increases in fire frequencies, further fragmentation of native communities, and declines in obligate shrub steppe wildlife species (Shaw et al. 1999). As a result, biodiversity was reduced and portions of the shrub steppe are now in transition to annual grasslands. Considerable information is available on the biology and ecology of cheatgrass and medusahead communities. However, effective and reliable measures have not been developed for establishing fuel breaks or for reestablishing native communities in annual grass-infested areas (Young and Evans 1970, Monsen 1994, Monsen and McArthur 1995, Roundy et al. 1997). Control of annual grasses to enhance revegetation requires depletion or removal of the annual-grass seed bank before or during the seeding operation. Techniques for accomplishing this include spring tillage or spring burning to eliminate seed production during the current year. Fall tillage is required if residual seeds germinate following autumn precipitation. Effectiveness of these techniques varies widely, depending on terrain, local weather conditions, treatment timing relative to cheatgrass or medusahead development, and recovery of annuals from residual seed reserves left near the soil surface where they are capable of germinating (Stewart and Hull 1949, Hull and Holmgren 1964, Klomp and Hull 1972, Young and Allen 1997).

Public concerns regarding pesticide use, legal restrictions on herbicide use on public lands, and a lack of chemicals effective for controlling annual grasses but of low toxicity to nontarget species generally precluded use of herbicides to control cheatgrass or medusahead until approval of the "Final Environmental Impact Statement for Vegetation Treatment on BLM Lands in Thirteen Western States" (USDI 1991), which permitted the use of 21 herbicides on rangelands. One of these herbicides, OUST® (sulfometuron methyl), a sulfonylurea, is an effective pre- and post-emergent herbicide when applied at low levels (<70 g/ha[a]). OUST® acts on Photosystem II of photosynthesis, where it inhibits acetolactate synthase, an enzyme that catalyzes the production of branched-chain amino acids. Its action is most pronounced in actively growing meristematic tissue. Thus, although it will kill germinating annuals and perennials at low application rates, its impacts on established perennials are generally minor (DuPont 1996, 1997). Sensitivity, however, varies among species, and resistance can develop following repeated treatments (Blair and Martin 1988). OUST® has low toxicity and does not accumulate in animals (DuPont 1996, 1997).

OUST® is a granular, water-dispersible herbicide that provides general weed control (DuPont 1996). In Idaho it has been labeled for aerial application (helicopter) to control cheatgrass on noncropland (DuPont 1997). The recommended application period is within 6 weeks of soil freezing in fall or less than 6 weeks after soil thaw in spring. Water is required to move OUST® into the root zone where it acts as a pre-emergent through root uptake by germinating seedlings. Post-emergence control results from uptake through both roots and leaves. The half-life of OUST® in soil is 20 to 100 days; it is generally more persistent in alkaline soils, at cold temperatures, and in dry situations (DuPont 1996, 1997). Revegetation on OUST®-treated sites must, therefore, be delayed until the succeeding fall. In addition, grazing must be deferred for 12 months following treatment (DuPont 1997).

Initial Bureau of Land Management (BLM) OUST® applications following wildfires in the early 1990s indicated promise in controlling cheatgrass and improving the opportunity for successful establishment of revegetation species (Pellant et al. 1999). Deteriorating conditions in the Snake River Birds of Prey National Conservation Area (NCA) following large fires in the mid-1990s (USDI 1996, Shaw et al. 1999) generated interest in using OUST® at a larger scale to facilitate revegetation

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[a] Metric conversion: 70 g/ha = 1 oz/ac

of these critical areas. However, a number of questions remained relative to effective application rates and dates, the longevity of OUST® in soil, and its effects on biological soil crust species, nontarget vascular plants, and common revegetation species seeded in fall after treatment of burned or nonburned sites. Preliminary data from an ongoing study presented here describe the effects of date and rate of OUST® herbicide application on control of cheatgrass and medusahead on nonburned sites.

Study Areas and Methods

Field testing was conducted at 2 annual grass-dominated sites in the sagebrush steppe zone of southwestern Idaho (Table 1). The Orchard Research Site (Orchard) is about 32 km southeast of Boise, Idaho, and the Lucky Peak site, about 25 km east of Boise. The USDA Agricultural Research Service monitors precipitation and air temperature continuously at Orchard. Weather data are recorded by the USDA Forest Service at the Lucky Peak Nursery about 6 km northwest of the Lucky Peak site.

A grid of 4-m x 4-m plots with 1-m borders was installed at each site. OUST® was applied using a hand-operated, small-plot precision sprayer. Treatments were date and rate of application. Individual plots were treated either in fall 1996 (November 6) or spring 1997 (April 8). Application rates at Orchard were 0, 17.5, 35, 52.5, 70, 87.5, or 105 g/ha. At Lucky Peak these rates were doubled due to greater accumulation of litter and standing dead grass on the medusahead site (3,030 compared to 2,308 kg/ha in fall 1996 and 2,105 compared to 1,807 kg/ha in spring 1997). Rates, therefore, were 0, 35, 70, 105, 140, 175, and 210 g/ha. There were 5 replications of each treatment combination.

Annual grass production was determined by harvesting and drying grasses at peak production (May 29 to June 2, 1997, at Orchard; July 3-10, 1997, at Lucky Peak) from 2 0.1-m^2 subplots in each plot. Samples of seeds harvested from 3 additional subplots in each plot were used to estimate seed mass and density (200 seeds) and germination (50 seeds). Germination was tested by incubating seeds at 20°C/10°C (16 hrs/10 hrs) with exposure to cool-white fluorescent light during the high-temperature period. Seedlings were considered normal if all struc-tures essential for development were present and the radicle was 5 mm long. Firm, nongerminating seeds were tested for viability using tetrazolium chloride stain-ing (Moore 1985).

The effects of application date and rate on biomass production, viable seed mass, total percent germination, and density of viable seeds produced were examined using analyses of variance. The arcsin transformation was used to normalize percent data; count data were normalized using the square root transformation (Snedecor and Cochran 1980). Means were separated by Fisher's least significant difference. All differences reported were significant at $P < 0.05$.

Results

Orchard

Some cheatgrass seeds were germinating in the litter and surface soil on the fall 1996 application date (November 6). Soil water content was 8%. About 119 mm of precipitation was received between November 18 and 24. Air temperatures were below freezing at night but remained above freezing during the day until mid-December. Cheatgrass seedlings were in the 4- to 6-leaf stage on the spring 1997 application date (April 8). Thirteen mm of precipitation was received on this date with an additional 117 falling between April 17 and 23, 1997. Little rain was received in late April or May .

Control of cheatgrass production following OUST® treatments differed by application date. Compared to control plots, which produced 66 g/m^2 of cheatgrass, production was reduced by about 95% on fall-treated plots and by 60% on spring-treated plots.

Few seeds were produced on plots receiving OUST® treatments of 35 g/ha or greater in fall. Seed mass (1.5 ± 0.1 mg [S.E.]) and total percent germination (54 ± 4%) did not differ among spring treatments and controls. Production of viable seeds on plots treated at rates from 35 to 105 g/ha in spring was reduced by 90% compared to the control (324 compared to 2,863 seeds/m^2). Seed density on plots treated at 17.5 g/ha was intermediate and did not differ from the controls or from plots treated at higher rates.

Lucky Peak

Considerable medusahead seed germination occurred prior to the fall 1996 application date (November 6). Soil water content was 15% on this date. About 72 mm of precipitation was received between November 12 and 22, 1996. February and March were quite dry. Medusahead seedlings were in the 4- to 5-leaf stage on the spring 1997 application date (April 8). Soil water was 18%. About 48 mm of precipitation was received between April 18 and 23. As at Orchard, late April and May were dry.

Medusahead production varied with application date and rate. Control plots produced 126 g/m^2. Fall treatments at the lowest rates reduced biomass production by 93%; higher rates eliminated nearly all germinants. Controls and plots treated in spring at 35 g/ha produced similar amounts of biomass (113 g/m^2), while production on plots treated at rates from 105 to 210 g/ha (21 g/m^2) was reduced by 82%. Biomass on plots treated at 70 g/ha in spring was intermediate and did not differ significantly from either of these groups.

Plots treated in fall at rates of 70 g/ha or greater produced few seeds, but those treated at the 35 g/ha rate produced 1,611 seeds/m^2. Seed mass (4.7 ± 0.2 mg) and total percent germination (86 ± 3%) did not differ among the spring treatments and controls. However, spring treatments at 70, 140, 175, and 210 g/ha decreased seed density by about 80% compared to the controls (2,700 compared to 12,502 seeds/m^2). Seed density following the 35 and 105 g/ha spring treatments was intermediate and did not differ from the controls or from plots treated at the other rates.

DISCUSSION AND CONCLUSIONS

OUST® applied in fall at 35 g/ha or greater on cheatgrass and at 70 g/ha or greater on medusahead controlled annual grasses during the subsequent growing season. All spring treatments made after the grasses reached the 4- to 6-leaf stage were considerably less effective. In addition, yellowing and reduced growth were noted for some plants of native perennial bunchgrass species such as Sandberg bluegrass (*Poa secunda* Presl.) and bottlebrush squirreltail (*Elymus elymoides* [Raf.] Swezey) in plots treated with OUST® at rates of 70 g/ha or greater in spring. Thus, treatments in fall or possibly earlier in spring may be more satisfactory. Removal of litter and seeds by prescribed burning prior to OUST® application or treatment after wildfires may reduce the application rate required for uniform and effective control of annuals, particularly on medusahead sites. Although some fall treatments controlled annual grasses during the subsequent growing season, environmental conditions in the treatment and subsequent year will affect the longevity of OUST® in the soil, germination of annual grass seeds, and growth and seed production of annual grass seedlings. These factors, in turn, will affect the establishment of revegetation species. Final results of the current study will improve our understanding of OUST's® potential for aiding in the restoration of shrub steppe communities.

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Table 1. Description of OUST® Study Sites

Characteristic	Orchard Research Site	Lucky Peak
Location	Lower Snake River Plain, southwestern Idaho	Boise Front, southwestern Idaho
Native vegetation	Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* [Beetle & Young]) Sandberg bluegrass Thurber needlegrass (*Achnatherum thurberianum* [Piper] Barkworth)	Wyoming big sagebrush Antelope bitterbrush (*Purshia tridentata* [Pursh] DC) Bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Love)
Disturbance vegetation	Cheatgrass	Medusahead
Elevation (m)	970	975
Slope (°), aspect	0-2, variable	8-15, east-southeast
Mean annual precipitation (mm)	200-300	430
Mean annual frost-free days	140-190	126
Soil type	Sandy, mixed, mesic Xeric Torriorthent	Fine, montmorillonitic, mesic, Cumulic Haploxeroll

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EXPLORING THE POTENTIAL FOR BIOCONTROL OF CHEATGRASS WITH THE HEAD SMUT PATHOGEN

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Restoration of wildlands infested with cheatgrass (*Bromus tectorum* L.) through direct seeding of native species is hampered by competition from cheatgrass during the seedling establishment phase. Methods currently available for cheatgrass control include early-season burning, tillage, and herbicide application. These methods are sometimes either hazardous, expensive, or disruptive to remnant perennials and soil and may also be of limited effectiveness, depending on weather and other factors such as the roughness of the terrain. A novel approach for control of cheatgrass in conjunction with restoration seedings would be to use a naturally occurring pathogen, the fungus *Ustilago bullata*, which causes head smut disease, as a biocontrol agent. We are currently engaged in exploratory research to determine whether this approach is feasible.

The head smut pathogen infects at the seed germination phase of the cheatgrass life cycle. Its spores germinate along with cheatgrass. After a period of yeast-like proliferation as sporidia in the haploid stage, sporidia of opposite mating types fuse to form infection hyphae that penetrate the emerging coleoptile (Fischer and Holton 1957). The fungal mycelium ramifies through the plant during vegetative growth, but few or no symptoms of infection are evident. When the plant shifts to its flowering mode, the pathogen takes over the physiology of reproduction and head smut spores are produced instead of seeds. These are dispersed along with the seeds produced by adjacent healthy plants, and the cycle continues when spores and seeds once again germinate together.

The head smut pathogen has a wide host range and can infect a variety of native and introduced grass genera in addition to most species of *Bromus*. These include *Agropyron, Elymus, Hordeum*, and *Festuca*. But specific strains of the head smut pathogen are specialized onto specific hosts so that any 1 strain has a very limited host range (Fischer 1940). This will make it possible to use this pathogen as a biocontrol agent against cheatgrass with little or no danger to native hosts or crop species.

The head smut pathogen can be found in virtually all populations of cheatgrass, at infection levels that vary from only a few percent to nearly 100%. The first question to ask when considering such a ubiquitous pathogen as a biocontrol agent is whether it can push host populations to extinction. In order to be useful for biocontrol, it would be necessary to trigger an epidemic of such severity that cheatgrass seed production would be reduced to near zero. In general, such disease levels would be counter-adaptive for the pathogen, as there would be no seeds produced to provide hosts for infection the following year. But head smut spores are highly dispersable, so they could conceivably travel to new cheatgrass patches outside the epidemic area. In fact, there are anecdotal reports of head smut epidemics that temporarily eliminated cheatgrass over considerable areas (Fleming et al. 1942; Stewart and Hull 1949; Ralph Holmgren, USDA Forest Service retired, personal communication). We have recently observed a similar phenomenon near Arrowrock Dam in southern Idaho. Without follow-up seeding of desirable species, such areas are occupied by other weeds or are reinvaded by cheatgrass over time.

Our approach to the study of biocontrol potential in this pathosystem is to look at factors that might limit infection levels and to find ways to overcome these limits. We propose 3 hypotheses for why head smut disease epidemics may be limited:

1) The first explanation is that inoculum levels might be limiting, even when infection levels were high the previous year. In order to infect successfully, a spore must be located in a strategic position relative to the emerging coleoptile. The higher the spore density, the more likely it is that 1 or more spores will be in this strategic position. If spore density is limiting in the field, it might be possible to cause severe epidemics just by adding more spores.

2) A related hypothesis is that there are multiple races of head smut in a population and each is pathogenic on only a subset of cheatgrass plants. Because

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cheatgrass is obligately inbreeding, different individuals in a population are either very closely related (members of the same inbred line) or only distantly related (members of different inbred lines) (McKone 1985, Novak et al. 1991). We hypothesize that different inbred lines of cheatgrass in a population are susceptible to different strains of the pathogen (i.e., there is resistance polymorphism). If this is the case, one can envision a scenario where different inbred lines within a population "take turns" being smutted, a phenomenon known as frequency-dependent selection. If a smut strain is abundant, it will infect most members of susceptible inbred lines, causing those lines to become rare. This in turn will cause that smut strain to become rare, which will permit those inbred lines to once again become abundant. Thus the cheatgrass individuals that are abundant in a given year would be the progeny of lines that were resistant to the abundant smut strain the previous year. This scenario puts a new twist on the idea of limiting inoculum level, because not only would spores have to be in the strategic position for infection, they would also have to be the right spores – the ones that are pathogenic on that particular inbred line. The net effect would be to make severe epidemics less likely, because the pathosystem would achieve dynamic equilibrium. The way to disrupt this equilibrium for purposes of biocontrol would be to apply inoculum in which all smut strains are equally abundant.

3) A third hypothesis for why head smut disease epidemics are rare is that environmental conditions required for infection are not met every time a cheatgrass seed germinates, i.e., just having the right spore in the right place is not sufficient. Cheatgrass seeds are more or less dormant at dispersal in early summer but become completely germinable over a wide range of temperatures by fall (Meyer et al. 1997, Bauer et al. 1998). Not all the seeds germinate at once, however. Timing of germination depends on both rainfall and microsite, which interact to determine whether a seed stays wet long enough to germinate. In addition, based upon drying rate after a wetting event, the seeds may or may not "remember" their previous wetting experience and germinate more quickly the next time they are wetted (Allen et al. 1993, Debaene-Gill et al. 1994). In order to infect successfully, the spores of the pathogen must track the germination timing pattern of the seeds. If the spores adjacent to a seed germinate either too soon or too late, they will miss the window of infection opportunity.

In order to test these complementary hypotheses and arrive at a procedure for causing head smut disease epidemics in the field, we will have to carry out a complex series of studies. Many of these studies are already in process. One thing we have learned is how to get reliable infection of susceptible plants in a greenhouse setting, i.e., when we inoculate the seeds of susceptible lines, the resultant plants consistently grow up smutted.

We have good evidence now that there is resistance polymorphism, both among populations and within populations. For example, cheatgrass plants from a southern Nevada population were resistant to pathogen populations from northern Utah but were completely susceptible to their own pathogen population. The converse was not true, however. When we inoculated northern Utah cheatgrass populations with spores from the southern Nevada pathogen population, most of the plants were as susceptible to that pathogen population as they were to their own pathogen population. We used bulk inoculum representing all strains of a pathogen population in these tests.

We then isolated monosporidial lines from a pathogen population, grew them in culture, and used paired lines of opposite mating types to inoculate cheatgrass seeds. Instead of the pattern of relatively high infection for all cheatgrass inbred lines that we obtained with the bulk inoculum, we found that some lines within a population were highly resistant to the isolate pair used, while the rest were highly susceptible. This is strong evidence for resistance polymorphism within populations, providing support for the "inbred lines take turns being smutted" hypothesis. We plan to pursue this technique systematically, with the goal of developing resistance-pathogenicity matrices for selected cheatgrass populations. These matrices will describe all known pathogen races for a cheatgrass population in terms of which inbred lines each can infect.

Another approach for testing the "inbred lines take turns being smutted" hypothesis involves use of molecular genetic markers for identifying cheatgrass inbred lines from tissue samples collected in the field. We have developed a set of 5 microsatellite marker loci that potentially permit us to distinguish among over 300 genotypes. We have identified DNA microsatellite fingerprints using variation at these 5 loci for a representative sample of individuals from each of 4 cheatgrass populations and have identified the inbred lines within each population. We also collected tissue samples in 1999 from 100 smutted plants and 100 unsmutted plants in each of 2 populations. By looking at the frequency distribution of inbred lines in smutted and unsmutted categories, we can learn whether any inbred lines are over-represented in the smutted category. And by carrying out this type of analysis for at least 3 years, we can find out whether inbred lines do indeed "take turns" being smutted.

To address the question of limiting environmental conditions, we have carried out a series of greenhouse experiments aimed at defining optimum conditions for infection. We have looked at the effects of temperature, fertility, soil pH, planting depth, litter type, and vernalization treatment on infection success. In addition, we are examining the possibility that different pathogen populations have different requirements regarding the window of opportunity necessary for infection. For example, we have found that the spores of the pathogen, like the seeds of the host, are more or less dormant at maturity and lose dormancy under dry conditions. It may be that pathogen populations from contrasting environments differ in patterns of after-ripening as a function of temperature, as is the case for cheatgrass populations (Meyer et al. 1997, Meyer and Allen 1999). We also have some very preliminary evidence suggesting that different pathogen populations have different temperature ranges and optima for spore germination, sporidial proliferation, and successful coleoptile penetration. If ecotypic variation of this kind is found in the pathogen, it may be possible to make use of a pathogen population with less specific requirements for infection, a broader infection window, and consequently higher infection success.

We are still a long way from developing a biocontrol procedure based on head smut disease for use on the ground, but our results to date make us optimistic that biocontrol of cheatgrass using the head smut pathogen may become a practical option. The time frame for application would be similar to that used for the pre-emergent herbicide OUST®. Fall application prior to cheatgrass emergence would cause smutting and prevent seed set in the following spring so that restoration species could be seeded in the fall 1 year after spore application. The head smut biocontrol method would also provide some follow-up control in subsequent years on plants that come from seeds that disperse in from adjacent areas or have persisted across years in the soil seed bank.

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USE OF NATIVE PLANTS FOR SAGEBRUSH STEPPE RESTORATION

T.A. Jones

WHY NOT NATIVES?: A BRIEF SUMMARY OF HOW CRESTED WHEATGRASS CAME TO PREDOMINATE CONSERVATION SEEDINGS IN THE INTERMOUNTAIN REGION

For many years, introduced species such as crested wheatgrass (*Agropyron* spp.) have been used to reseed sites like the Snake River Birds of Prey National Conservation Area (NCA). While the interest level for native species has greatly increased in recent years, introduced grasses continue to be predominant in seedings in the region. The rise in use of crested wheatgrass in the semiarid West resulted from a large research effort in the mid-1900s following failed efforts to seed native grasses earlier in the century (Sampson 1913). Here I will explore the linkage between crested wheatgrass's rise to prominence and its timely ability to meet the societal needs of its day at both local and national levels. Those critical of the values of past generations should remind themselves of the historical context within which land management decisions were made (Roundy 1999).

Invasion of Intermountain rangelands with exotic weeds following their introduction in imported lots of crop seeds and ship ballast, coupled with overgrazing, was noticeable by the end of the 19th century. Shortly after the turn of the century, Kennedy and Doton (1901) and Griffiths (1902) began calling for restoration of degraded big sagebrush (*Artemisia tridentata* Nutt.) shrublands by seeding native perennial grasses. Despite research efforts, development of technology to seed grasses, either introduced or native, on semiarid lands did not succeed at this time. Jardine and Anderson (1919) concluded that the only Intermountain environments that could be economically seeded were alpine sites. By 1930, A.W. Sampson had given up on his early efforts to seed native grasses to curb watershed deterioration in the Wasatch Mountains of central Utah (S.B. Monsen, USDA Forest Service, personal communication). Smooth bromegrass (*Bromus inermis* Leyss.), a rhizomatous introduction with excellent forage production and erosion-controlling ability, was used instead.

Crested wheatgrass was first introduced to the United States by N.E. Hansen of the USDA South Dakota Agricultural Experiment Station in 1898 (Rogler and Lorenz 1983). There are no records of any seed increases or performance data from this material. Later introductions were made in 1906 and 1908. While interest in this grass was primarily confined to the Dakotas, seedings were made at Union, Oregon; Moro, Oregon; and Moccasin, Montana (Jackman et al. 1936, Dillman 1946). Seed was increased at Moro and distribution was made for research purposes (Dillman 1946). This early success did not lead to an immediate widespread adoption of crested wheatgrass. Large tracts of Intermountain rangelands were plowed and seeded to wheat and barley as World War I increased worldwide demand for grain due to decimation of European grain production. The expansion of rangeland grain acreage during the war led to collapse of inflated grain prices during the 1920s. With the advent of several years of drought and wind erosion on lands unsustainable for grain production in the "Dirty Thirties," the farm economy was weakened considerably. As the Great Depression deepened, many agricultural lands were abandoned and fell into government hands in lieu of delinquent taxes.

By this time, crested wheatgrass seed had become readily available for the first time. Government agencies like the Civilian Conservation Corps used hand seeders to seed either around sagebrush plants or in monocultures subsequent to sagebrush removal with tractor-drawn rails (Young and Evans 1986). These hand-seeding efforts were generally failures, but as mechanized agricultural techniques came into general use in the late 1930s, seedings became more successful.

But again, international circumstances circumvented well-meaning attempts to improve ecological status of western lands. World War II created a new demand for red meat and led to large increases in livestock numbers, particularly sheep, in the West. In light of the seeding failures of the 1930s, Congress appropriated money to fund pilot programs to research crested wheatgrass seeding on public lands in Utah, Idaho, and Nevada. These programs were headed by A. Perry Plummer, A.C. Hull, and Joseph H. Robertson, respectively. Instead of restoring native rangelands, the prevailing concept at this time was to establish productive pastures of introduced grasses to take spring grazing pressure off the remaining native grasslands. In this manner, policy makers hoped to control soil and water erosion, maintain ecological status

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of native grasslands, and meet the wartime demands for red meat production.

While it was recognized that crested wheatgrass had potential in the Intermountain Region as well as the Great Plains, researchers still needed advances in tillage and seeding equipment (Hull and Pearse 1943, Plummer et al. 1943, Robertson and Pearse 1943). The Forest Service Equipment Laboratory (Portland, Oregon) developed the brushland plow, which was able to remove sagebrush in rocky soils. Based on an Australian prototype, the implement featured independently suspended disks on spring-loaded arms to eliminate equipment damage caused by rocks. In 1951 a prototype rangeland drill was unveiled (Young and McKenzie 1982). Rangeland drills featured "depth bands" mounted on the coulters to permit precise furrow depth and seed placement (1/4" to 1/2" [6-13 mm]) despite unevenness of the seedbed. Press wheels firmed the seedbed against the seed, improving the probability of seedling establishment.

These research advances coincided with a new rangeland crisis. In 1947 large numbers of sheep died in the Intermountain Region (Matthews 1986). Eventually, oxalate poisoning by halogeton (*Halogeton glomeratus* [Bieb.] C.A. Mey), a new range weed, was pinpointed as the causal agent. This previously unrecognized weed quickly became a threat to livestock agriculture in the region, by this time a well-established industry. Political clout by livestock interests led to the passage of the Halogeton Control Act of 1952. The Act provided federal funding for extensive crested wheatgrass seedings by the Bureau of Land Management. By this time, Joe Robertson's research plots near Wells, Nevada, had shown that crested wheatgrass could suppress invasion by halogeton, much as this perennial grass is used to suppress cheatgrass in the NCA today. Crested wheatgrass eliminated halogeton as a range problem and earned crested wheatgrass the moniker "Saviour of the West." The realization that crested wheatgrass was palatable to livestock when planted in monocultures, despite its nonpreference in mixed stands, added to its reputation within the livestock community.

These successes ushered in the "Golden Age of Rangeland Seeding," a period of about 10 years when extensive seedings of crested wheatgrass were established (Young and Evans 1986). This occurred because comprehensive research had been completed on both the plant and planting methodology, excellent communication was established between researchers and implementers, and economic conditions favored investment in industry following World War II. This trend continued until environmental consciousness raised vociferous objections to monoculture seedings of introduced species in the mid-1960s, continuing to the present day. Since that time, seeding of crested wheatgrass has continued, albeit at a reduced rate. In the last 13 years, crested wheatgrass has been widely established in mixed stands on private land because of demand created by the USDA National Resources Conservation Service (NRCS) Conservation Reserve Program (CRP). The success of the cultivar Hycrest, released in 1984, was partly due to its coincidental concurrent appearance in the marketplace with the onset of the CRP.

How natives are being implemented now: an example from the Forest Service, Region VI

Among the native perennial grasses, those with cultivars with good seed production have been able to significantly impact the market, e.g., green needlegrass (*Nassella viridula* [Trin.] Barkw.; first release, 1946), slender wheatgrass (*Elymus trachycaulus* [Link] Gould ex Shinners; 1946), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] A. Löve; 1946), thickspike wheatgrass (including streambank wheatgrass) (*Elymus lanceolatus* [Scribn. & J.G. Smith] Gould; 1954), western wheatgrass (*Pascopyrum smithii* [Rydb.] A. Löve; 1970), Indian ricegrass (*Achnatherum hymenoides* [Roem. & Schult.] Barkw.; 1974), basin wildrye (*Leymus cinereus* [Merr.] A. Löve; 1979), and Snake River wheatgrass (*Elymus wawawaiensis* J. Carlson and Barkw.; 1980). Misconceptions that cultivars contain less genetic diversity, discomfort with their seeding over large expanses, and concern that their adaptation may be inferior to native-site material have encouraged some to explore seed increase of site-specific material. Here I will review a recent effort by Vicky Erickson and her colleagues in the Umatilla and Hood River National Forests (Pendleton, Oregon), Forest Service Region VI, and the commercial seed sector (http://www.fs.fed.us/r6.uma/native). Similar efforts have been effected by cooperation between the National Park Service and the NRCS (Beavers 1995, Lange and Lapp 1999, Majerus 1999).

Forest Service personnel made seed collections within subwatersheds and 500-foot (152-m) elevation bands. For seed-increase purposes, collections were bulked up to the watershed and 1,500-foot (456-m) elevation band level. In 1993, the J. Herbert Stone Forest Service nursery (Medford, Oregon) contracted to increase bulked seed of 6 grass species in 2.5-acre (1-ha) fields. The Stone nursery has continued seed increases of species with problematic seed production along with material newly entered into the program (Steinfeld 1997). But beginning in 1996, a Washington seed company has won 3-year contracts awarded on an annual basis to increase seed of the program's "workhorse species," i.e., blue wildrye (*Elymus glaucus* Buckl.) and mountain brome (*Bromus carinatus* H. & A.). These grasses are good seed producers and early seral in their successional status, making them widely useful in fire rehabilitation. They eventually yield to longer-lived native species if seed of the latter is present in the soil. The Forest

Service specifies 1) a quantity estimate and 2) a maximum right-to-purchase quantity with bidders responding with a price. The program currently entails 13 native grass species and 25 acres (10.1 ha) of seed production with an additional 12 acres (4.9 ha) to be established in the fall of 1999. This program has been encouraged by a 1995 Region VI policy promoting natives in reseedings, falling prices as start-up costs decline, and a ready market for seed beyond Forest Service needs created by the CRP.

Besides the temporary nature of the CRP, several factors may limit the long-term endurance and expansion of the program. Because seed needs cannot be forecast for fire rehabilitation as can be done for forest trees for timber-cut reforestation, it is difficult to request intramural funds for this work, given present bureaucratic procedures. Unlike conifer seedling production, stable year-to-year funding is not in place for herbaceous seed production in the Forest Service. Finally, lack of environmentally controlled warehousing facilities makes it difficult to accumulate seed stocks for anticipated future needs. Current federal agency priorities are responsible for the favorable climate toward native-site seed production, but future funding constraints may limit its expansion, just as historical circumstances favored the adoption of crested wheatgrass and other introduced grasses throughout much of the 20th century.

BEYOND "NATIVES": THE RESTORATION GENE POOL CONCEPT AS AN ALTERNATIVE TO THE NATIVE/NONNATIVE DICHOTOMY

The restoration gene pool (RGP) concept defines 4 ranked gene pools (primary, secondary, tertiary, and quaternary), with the primary RGP preferred when it both "works" practically and promotes management objectives, and lower-ranking RGPs suitable when higher-ranking pools either do not "work" or do not meet management objectives. Degree of correspondence between genetic identity of the gene pools and the "target" native-site population declines from primary to quaternary RGP. The primary and secondary RGPs encompass the same taxon as the target, while the tertiary and quaternary RGPs are of distinct taxa. The primary RGP includes only material from the target site or adjacent connected areas (the metapopulation). The secondary RGP originates from genetically disconnected sites of the target taxon. Tertiary RGP taxa have been intimately connected to the evolution of the target taxon, but modern gene flow is precluded by a genetic barrier. The quaternary RGP involves taxa that are at most remotely involved in the evolution of the target taxon but that provide similar ecosystem structure and function, including introduced species.

For bluebunch wheatgrass with the NCA as target site, the secondary RGP consists of material originating from various disjunct (genetically disconnected) sites. Examples include the multiple-origin polycross P-7 (USDA-ARS, Logan, Utah) (Larson et al. In Press) and single-site populations such as "Whitmar" (Colton, Whitman Co., Washington), "Goldar" (Mallory Ridge, Umatilla N.F., Asotin Co., Washington), B53 (Anatone, Asotin Co., Washington; USDA-FS, Provo, Utah), and Acc:238 (Lind, Adams Co., Washington; USDA-ARS, Logan, Utah). I will be pursuing discussion of Acc:238 because natural average annual precipitation at Lind, Washington (9.5" [241 mm] over 69 years), is similar to parts of southwestern Idaho (http:/www.ftw.nrcs.usda.gov/prism/prism.html; http://www. ncdc.noaa.gov/pub/data/coop-precip/washington.txt). Winter hardiness zone at Lind (6a) is also comparable to southwestern Idaho (5b-7a) (http://www.ars-grin.gov/ars/Beltsville/na/hardzone/ushzmap.html). But unlike bluebunch wheatgrass germplasm originating in southwestern Idaho, Acc:238 exhibits good seed production, making it a practical alternative to the primary gene pool. This demonstrates that despite its lower genetic identity to the targeted population, the secondary RGP may still be highly adapted.

In the case of bluebunch wheatgrass, the barrier between secondary and tertiary RGPs is polyploidy. While southwestern Idaho bluebunch wheatgrass is diploid ($2n=14$), many populations from more mesic environments in the Northwest are tetraploid ($2n=28$). Although no flora actually recognizes tetraploid bluebunch wheatgrass as a separate taxon, this example works well conceptually because of genetic isolation between diploid and tetraploid populations. Even when direct use of the tertiary gene pool is inappropriate because of poor adaptation or genetic sterility barriers, its germplasm may be useful to plant materials researchers for developing adapted material via conventional techniques of artificial hybridization and chromosome doubling.

The quaternary RGP, in contrast to the tertiary RGP, involves taxa at most remotely involved in the evolution of the taxon of interest but exhibits ecosystem structure and function reminiscent of the primary gene pool and meets management objectives when implementation of primary, secondary, or tertiary RGPs is problematic. The quaternary RGP for bluebunch wheatgrass at the NCA includes other *Pseudoroegneria* species (all introduced), Snake River wheatgrass, and crested wheatgrass. Several species of *Pseudoroegneria* have been introduced from central Asia (http://www.ars-grin.gov/cgi-bin/npgs/html/tax_search.pl?Pseudoroegneria). Snake River wheatgrass, e.g., "Secar" (Carlson and Barkworth 1997), is a grass not native to the NCA that has been successfully introduced to this target site.

Crested wheatgrass has long been utilized as a corollary to bluebunch wheatgrass in the Intermountain Region because it has met the management objective of forage for livestock grazing. Moreover, like bluebunch wheatgrass, it is adapted to similar climatic regimes and exhibits a caespitose growth habit, but it is more competitive and tolerant of grazing (Caldwell et al. 1981, Richards and Caldwell 1985, Mueller and Richards 1986, Busso et al. 1989). Crested wheatgrass's genetic identity is dissimilar to bluebunch wheatgrass, but crested wheatgrass's adaptation is very high in the NCA ecosystem, particularly since it has been perturbed by annual weed invasion and unnaturally high fire frequency. There is a place for introduced species in restoration and it is in the quaternary RGP.

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HOW TO GET THE NATIVE SEED YOU WANT: A PRODUCER'S PERSPECTIVE

Claire Gabriel Dunne

Some species of native seed are available. The Conservation Reserve Program has increased the demand for certain native species such as:

thickspike wheatgrass (*Elymus macrourus*)
bluebunch wheatgrass (*Pseudoroegneria spicata*)
slender wheatgrass (*Elymus trachycaulus*)
streambank wheatgrass (*Elymus lanceolatus*)
western wheatgrass (*Pascopyrum smithii*)
big bluegrass (*Poa ampla*)
Canby bluegrass (*Poa canbyi*)
Sandberg bluegrass (*Poa secunda*)
big bluestem (*Andropogon gerardi*)
little bluestem (*Schizachyrium scoparium*)
blue grama (*Bouteloua gracilis*)
sideoats grama (*Bouteloua curtipendula*)
Indian ricegrass (*Achnatherum hymenoides*)
green needlegrass (*Nassella viridula*)
prairie sandreed (*Calamovilfa longifolia*)

In response to the huge demand, native seed growers have increased acreage. Although prices are high, they will drop precipitously in the lag time between the end of the Conservation Reserve Program and the time growers start to plow under their stands due to low prices. You can buy seed by inviting reputable seed companies to offer you a bid, preferably requesting weight in Pure Live Seed (PLS).

If the species you desire is not commercially available, you can collect the seed yourself. Carefully note the location of the site, since it will be much harder to find out of bloom. You may need to revisit the site two or three times to determine ripeness. The rule of thumb is that seed ripens four weeks after bloom. When most of the seed is nearly ripe, take samples from several plants and cut at least 25 seeds to determine "fill." Filled seed contains a "nutmeat." Learn what the seed looks like from books or by experience, and use a hand lens to inspect the seed. Early in my career I collected a whole bag of anthers from shrubby cinquefoil (*Potentilla fruticosa*)! Try to harvest seed before it is removed by wind, insects, or feathered or hoofed predators. Fresh seed is damp; store it only in porous bags or boxes. Most seed must be spread out and stirred a few times a day until air dried, otherwise the material will heat up or mold. The last major hurdle is avoiding damage during seed cleaning. Many seeds are delicate and can be damaged by the improper application of horsepower and steel.

An increasing hazard to wildland collecting is the inexorable spread of noxious weeds thoughout the land. Many former collecting areas must be avoided by conscientious collectors because the risk of noxious weed contamination is too high. For example, knapweed species (*Centaurea* spp.) produce seeds borne on parachutes that float in the air and hang up in the nearby foliage of snowberry (*Symphoricarpos albus*). Learning the appearance of noxious weeds in your area in their dried condition will help avoid collecting from a contaminated patch.

If the process of collecting is daunting, you can contract a custom collection from a professional seed collector. It is likely that a collector will be able to gather enough seed for your test plot or to have grown into plants in a nursery. On the other hand, collecting enough to direct sow over even a few acres may be a herculean and expensive task. In the arid West, for example, most native species produce seed only once every 7 to 10 years. The second factor limiting collection is the scarcity of large, homogeneous, flat, accessible stands of native plants. It is common for collectors from several states to converge on a good patch of seed with mechanized equipment.

If you desire a reliable supply of certain species year after year, the best bet is to contract field production. There is inevitably some inadvertent selection of certain traits in the field; e.g., the earliest and last seed to ripen are not harvested, thereby selecting for a narrower bloom and ripening period. On the other hand, field production controls many of the variables needed to grow good seed, such as moisture, competition for nutrients and light, ungulate grazing (though insect and wildlife grazing can be serious), and timely harvest. Perhaps the most important is noxious weed control. Inspectors from the Crop Improvement Association will check not only for weeds in the crop field, but also for noxious weeds in ditch banks or other nearby areas.

To increase the chances of having seed available when your project requires reseeding, contract a minimum

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of 3 years ahead, though some species may take 10 years to fruition. If you can give the farmer enough seed to sow several acres the first planting season, he can multiply the stock seed by Year 2 or 3. However, if he has only enough seed to plant a short row, he will produce enough in Year 2 to plant 0.25 acre (0.1 ha) in Year 3. That quarter acre will come to fruition in Year 5 for sowing of several acres in Year 6. Thus, you won't see large-scale production until Year 8, assuming all else went well and none of the crop was lost along the way to hail, rabbits, flood, the canal breaking in the heat of the summer, or other perils of agriculture.

If there is a large, stable demand, prices are lower and farmers can use economies of scale to supply large quantities at lower unit prices. If land managers promise to order, say, 10,000 pounds (4,500 kg) of Idaho fescue *(Festuca idahoensis)* per year, several farmers will rise to meet the demand by risking planting a crop. These are the aspects of agriculture: supply, demand, and risk.

Larger orders encourage cleaning facilities to invest in technology. Effective cleaning machinery such as variable-speed hammermills, debearders, fanning mills, aspirators, length separators, and gravity separators are essential. Frequently, characteristics of the seed obstruct our attempt to achieve high purity. Many shrubs, such as winterfat (*Krascheninnikovia lanata)*, are cleaned to only about 80 percent purity due to the fuzz attached to the utricle. Removing this fuzzy utricle reveals a delicate nutlet which loses viability if not sown promptly. Also, the hair on the "pods" may aid germination by gluing the seed to the soil or by absorbing moisture. With these constraints in mind, the best an experienced conditioner can provide is large bags of fluffy utricles with stems removed. Conditioners are quite inventive: the late Roger Stewart built a "cannon" to shoot the product across the warehouse, allowing the heavier stems to fall short and the lighter utricles to be swept up and bagged.

Those of us in the private sector would like to see government facilities continue to provide research and development work, as have the Natural Resources Conservation Service (NRCS) Plant Materials Centers and the Agricultural Research Service (ARS) for decades. We find it hard to compete, however, with government-subsidized facilities such as the USDA Forest Service nurseries, which are growing native seeds. Unlike a farmer on private land, agency nurseries operate on public land, pay no taxes, and have free marketing and advertising. The more government agencies win production bids, the fewer natural resource graduates find work in the private sector. Private growers will be delighted, on the other hand, if public nurseries develop germination, production, and conditioning methods for useful species such as elk sedge (*Carex geyeri*) and pinegrass (*Calamagrostis rubescens*), thereby transferring the technology to private growers.

MANAGEMENT OF SAGEBRUSH STEPPE VEGETATION

Alma H. Winward

ABSTRACT

This presentation focused on the subspecies and varieties within the big sagebrush (*Artemisia tridentata*) group. The 6 or more taxa in this group are especially well adapted to the physical and environmental characteristics of the semiarid West. Various features of this group have allowed them to be especially competitive with their herbaceous understory. This includes features of their leaves, seeds, and roots, as well as special structural and physiological characteristics that allow them to be particularly dominant on portions of their ranges. Historic wildfires have played an important ecological role in allowing understory grasses and forbs to coexist with sagebrush. Changes have occurred between present and historical distribution, including reductions in acreages due to western agricultural practices, urbanization, and alterations in historical fire regimes. Functioning of sagebrush ecosystems can be altered by too much or too little fire compared to historical frequencies and patterns. Management practices needed to maintain these important ecosystems in the face of opposing interests were also discussed.

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ESTABLISHMENT OF BIG SAGEBRUSH (*ARTEMISIA TRIDENTATA*) IN SEMIARID ENVIRONMENTS

Stephen B. Monsen

INTRODUCTION

Seeding disturbed areas in semiarid sagebrush steppe and salt desert shrub communities is difficult and often unsuccessful. Jordan (1982) determined that low and irregular precipitation patterns in these environments were insufficient to sustain young seedlings in many years. Invasive annual weeds, principally cheatgrass (*Bromus tectorum*), medusahead (*Taeniatherum caput-medusae*), bur buttercup (*Ranunculus testiculatus*), various mustards (*Sisymbrium* spp.), and Russian thistle (*Salsola iberica*) provide sufficient competition to prevent seedling establishment of other species (Monsen 1994). These weeds are present in most sagebrush (*Artemisia* spp.) and salt desert shrub communities and restrict establishment of artificial seedings unless steps are taken to control weedy competition (Johnson and Payne 1968, Chatterton 1994). Few native or introduced species are capable of recolonizing disturbed big sagebrush (*Artemisia tridentata*) communities, particularly those dominated by competitive weeds. This paper discusses seed and seeding technology and microenvironmental requirements for the reestablishment of big sagebrush on such sites.

SELECTION OF ADAPTED SUBSPECIES AND ECOTYPES

Shultz (1986) reported that populations of big sagebrush display close alliance to certain habitats, and morphological specializations and adaptations have evolved along environmental gradients. This has produced the current distribution patterns of the principal subspecies of big sagebrush – basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*), mountain big sagebrush (*A. tridentata* ssp. *vaseyana*), and Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*). Davis and Stevens (1986) reported that significant differences in growth rates occurred within and among subspecies of big sagebrush grown in common gardens, indicating adaptation of these populations to their site of origin. Differences in photosynthetic characteristics among subspecies of big sagebrush described by Frank et al. (1986) also correlated with environmental conditions of their sites of origin. These and other studies suggest subspecies and ecotypes have evolved to survive in distinct environments; thus, movement of populations to locations with different climatic and edaphic conditions is not advisable.

CHARACTERISTICS OF BIG SAGEBRUSH SEED

Seed Production and Harvesting

Flowering occurs in late summer and early fall, and seeds mature in early winter. Drought, as well as late fall and early winter storms, coupled with persistent cold temperatures, can prevent seed development, diminish seed quality, and reduce seed harvests. Because plants are partially self-fertile, isolated shrubs do set seed (McArthur et al. 1988). Consequently, scattered shrubs growing with reduced intraspecific competition can produce large quantities of seed.

Factors including plant morphological characteristics, timing of flowering and seed maturation, and quantity of seed produced differ among big sagebrush subspecies and must be considered in their use. Basin big sagebrush plants normally grow on deep soils in valley bottoms where additional runoff accumulates. Individual plants are larger than those of mountain big sagebrush or Wyoming big sagebrush and produce greater numbers of flowers and seeds (McArthur and Welch 1982). In addition, seeds are more easily and efficiently harvested from the upright shrubs of basin big sagebrush. Consequently, accurate identification is essential to ensure that seed of this subspecies is not substituted for either Wyoming or mountain big sagebrush seed. Mountain big sagebrush plants usually produce a seed crop each year, but amounts are not always sufficient to justify harvesting. Plants of Wyoming big sagebrush are much less floriferous and produce little seed except in unusually wet years. Thus, crops are usually sparse and seed collection is slow and laborious.

Most commercial seed is harvested from wildland stands, and more favorable sites and subspecies are commonly harvested. Although many wildland stands do not yield harvestable crops each year, certain stands are consistently high producers and collectors protect and manage these areas. Seed produced under cultivation or in managed wildland stands can produce more consistent crops than most wildland populations (Wagstaff and

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Welch 1991). Young et al. (1989) concluded that seed production at the subspecies and population levels are, in part, regulated by genetic constraints. Thus, sites for seed fields must be carefully matched with the subspecies and ecotypes grown. Excessive irrigation and cultivation can be detrimental, resulting in diminished seed yields and plant vigor. Development of appropriate cultivation practices is essential as specific populations, including populations from arid regions where yields are naturally low, will likely be required to supply current demands.

Seed Cleaning

Large basin big sagebrush shrubs may produce as many as 500,000 seeds per year (Welch et al. 1990), but yields are normally much lower. Seeds of all subspecies are quite small with approximately 4.5 million per kilogram. As seeds mature, they slowly shatter and are dislodged by wind. Seeds may persist on the shrub for an extended period, sometimes permitting collection to be profitable for more than a month.

Seeds are harvested by stripping or flailing the flower stalks, which results in removal of some small branches, leaves, floral parts, and seeds. Seeds collected in fall and winter often have a high water content and must be dried prior to cleaning and storage. Water content of stored seed should not exceed 10%. In winter, forced-air drying in a heated building may be required to reduce the water content below this level.

Collected material is normally air dried, screened to remove large debris, and cleaned with a debearder to reduce bulk. Properly cleaned seed lots should be free of unnecessary twigs and other debris. Removal of this material is important to facilitate seeding, not simply to increase seed purity to a set percentage. In most cases, wildland-harvested big sagebrush seed is easily cleaned to a purity of at least 12%. If seed lots are free of unwanted sticks and related material and seed germination percentages exceed 75-80%, then a requirement for additional cleaning to attain higher standards is not necessary or practical. Repeated fanning and cleaning is generally required to increase the purity beyond 14-20%, thus increasing costs considerably.

Seeds should not be collected by clipping stems and floral stalks. When such material is cleaned with a debearder to separate seeds and floral tissue from the stems, small stem segments are broken and remain with the clean seed fraction. These short stems have rough, irregular ends that catch and interfere with the flow of material through seeding devices.

Using a pure live seed (PLS) basis to purchase seed is advisable to ensure that an adequate quantity of seed is purchased and to facilitate comparisons among seed lots. High PLS standards and restricted tolerance levels currently maintain seed prices at unnecessarily high figures. Inconsistent results from state seed testing laboratories contribute to these high prices. Suppliers are reluctant to sell seed, knowing test results are unpredictable and can result in significant penalties.

Cleaned seed lots can normally be planted with most seeding equipment. The material is not heavy and varies in consistency, which can reduce uniformity of flow through some seeders. Most problems can be corrected by using seed boxes with agitators. Seeds can be placed at desired depths in the soil without special attention if equipment is properly adjusted. Sagebrush seeds can be mixed with seeds of other commonly seeded species to plant at desired rates without adding carriers or diluents. Aerial seeding can be accomplished without any major problems.

Seeds are often collected in late autumn or winter after the planting season has passed. Consequently, a considerable amount of seed is stored for planting in subsequent years. Seed viability diminishes after 1 or 2 years, depending on storage conditions. Seed lots should be tested at the time seed is sold to ensure accurate results are provided to the buyer.

Seed Dormancy and Germination Patterns

Studies by Meyer and Monsen (1992) concluded that seed dormancy and germination patterns are habitat correlated among all three subspecies of big sagebrush and each subspecies exhibits a different pattern of variation. Habitat-correlated variation in germination appears to be an important adaptive characteristic of each subspecies. Seeds of mountain big sagebrush exhibited the greatest variation in dormancy and germination rates. Germination of these collections varied from 0 to 58%. Collections of mountain big sagebrush from severe winter sites required up to 113 days to reach 50% germination at cool temperatures. These collections contained a higher fraction of dormant seeds and germinated more slowly than collections of the other subspecies from mild winter sites. Basin big sagebrush and Wyoming big sagebrush seeds were mostly nondormant and germinated quickly at warm temperatures. Maximum percentage of dormant seeds was about 12% for basin big sagebrush and about 11% for Wyoming big sagebrush (Meyer and Monsen 1992). Collections of basin big sagebrush from mild winter sites required only 6 days to reach 50% germination. Although many environmental factors likely influence germination patterns, a close correlation existed between germination patterns and mean January temperature of the collection site.

Meyer and Monsen (1991) found that patterns of variation in big sagebrush seed germination are more strongly correlated with habitat conditions at the population level than with subspecies. These findings are important, as most seeds are usually nondormant at harvest. Primary dormancy and the light requirement of dormant seeds are removed by overwinter wet prechilling or dry afterripening. Consequently, differences in seed germination patterns among subspecies and ecotypes result from conditions seeds encountered between the time of dispersal and the optimum time for germination (Meyer 1994). Since seeds germinate over a wide range of temperatures, virtually all seeds from fall seedings germinate in spring. Studies conducted by Meyer and Monsen (1992) revealed that germination is regulated to coincide with conditions that favor seedling establishment. Movement of seeds from cold winter environments to more mild winter desert sites or reversing the exchange results in seeds germinating at less optimum periods. Thus, complete loss of entire seedings can occur if site-adapted sources are not planted.

Large amounts of seed are frequently purchased by the USDI Bureau of Land Management (BLM) and other agencies to plant following extensive wildfires. Use of seed acquired from a wide array of habitats obviously affects initial establishment and survival. Matching subspecies and populations with climatic conditions is certainly advisable. Matching big sagebrush seed with environmental conditions is more important for this species than for many other shrubs. Most big sagebrush seed is produced in autumn and either germinates or is lost from the seed bank by late spring. Only a fraction persists and emerges in the second spring. Meyer (1994) postulated that less than 1% of the seed may persist in the seed bank. Although this represents a sufficient number of seeds to establish a stand, only a few seedlings normally emerge in the second spring following seeding. Consequently, planting success of big sagebrush must be based on first-year establishment.

Natural seedling recruitment is required to maintain stands of big sagebrush. Poorly adapted sources may initially establish during certain years, but long-term survival of established stands relies on the ability of the planted population to recruit new plants over an extended period. Studies conducted in southern Idaho demonstrate that sources of basin big sagebrush acquired from central Utah and planted on basin big sagebrush and Wyoming big sagebrush sites in Idaho in 1978 and 1979 established dense and uniform stands on a variety of large tracts. Although established plants attained maturity and produced seed crops, sufficient new seedlings to maintain existing stands failed to establish by natural recruitment over the following 20 years. Loss of entire stands may not occur for 30-40 years unless some major event occurs to hasten the death of older, seed-producing plants. Consequently, matching site-adapted seed sources is essential to ensure initial seedling establishment and promote recruitment.

Field Germination Patterns

Autumn seedings of big sagebrush subspecies normally germinate in late winter and early spring. Young and Evans (1989) recorded the greatest number of seeds in a native seed bank in January, only a few weeks after dispersal. Based on studies established in central Utah, approximately 43% of planted seed germinated under a snow cover by late February. This represented about 57% of all seeds that ultimately germinated. Within 2 weeks, the snow had melted and all germination ceased. Germination of rubber rabbitbrush (*Chrysothamnus nauseosus*) followed the same pattern, although a higher percentage of seeds ultimately germinated. Seed germination patterns of both sagebrush and rabbitbrush progress much sooner and faster than many other shrub species. Germination of big sagebrush seed can occur from mid-winter to early spring, depending on weather conditions; but once conditions are favorable, rapid and complete germination can be expected.

Planting in well-prepared seedbeds frequently results in emergence of uniform stands. This may be desirable if favorable conditions prevail. It may also lead to complete failure under unfavorable situations. Seeding irregular surfaces by broadcast planting often results in less synchronized establishment patterns. This tends to enhance survival. Intraplant competition among big sagebrush seedlings is lessened, and seedlings are less condensed in rows or spots. Maintaining surface litter and irregular topography is also advisable to protect emerging seedlings and extend the period of establishment.

Large areas can be seeded with sagebrush quickly using aircraft. Autumn and mid-winter seedings are recommended to ensure seeds are in place when optimum conditions for germination occur. Seedings may be delayed until mid-winter to ensure optimum weather conditions exist to support germination. However, delaying seedings beyond mid-winter is not advisable.

Early spring germination favors emergence and seedling survival of big sagebrush plants. Seeds are able to germinate when soil water is most likely to be available and seedlings can better compete with associated vegetation. However, mid-winter and early-spring germination can also result in nearly complete elimination of all seedlings from rapid drying of the soil surface and periods of frost.

SEEDBED REQUIREMENTS

Seed Coverage

The small seeds of big sagebrush should be surface planted with only minimal soil coverage. Placing seeds at depths greater than 0.6 cm reduces emergence. Surface seeding on a firm but not compacted or crusted surface is more successful than drilling. Drill seeding can be successful if shallow depths can be maintained, but this is difficult to accomplish under rangeland conditions. Density of big sagebrush seedlings developing from broadcast, surface compaction, or drill seedings is often more variable than densities obtained from seedings of other shrub species. Differences in seeding density may result from irregular or uneven distribution of seed and irregular surface soil conditions. However, the overall density obtained in big sagebrush plantings is often greater for big sagebrush than for other shrubs. Plantings conducted in 1989 at the Solosabal and Poison Springs sites in southern Idaho were completed using similar equipment and methods. Considerable differences were recorded 1 year after planting, both within and between sites. Seedling numbers varied from 15,864 to 72,591 plants/ha at the more favorable Poison Springs site. In contrast, numbers at the more arid Solosabal site varied from 1,698 to 9,637 plants/ha.

Natural thinning of big sagebrush seedlings and young plants can occur over a 5- to 10-year period following planting. Approximately 42,000 seedlings and young plants/ha established and persisted for 2-3 years from seedings conducted in southern Idaho on Wyoming big sagebrush sites. Natural thinning occurred over almost 10 years until approximately 2,000 plants/ha remained.

Aerial seeding is highly successful if conducted in late fall and winter. Distributing seed on a rough but firm seedbed or on sites with surface litter often produces satisfactory stands. Broadcast seeding followed by light anchor chaining increased seedling density at the Dry Creek Drainage site in Idaho following a wildfire in 1992. Chaining to cover the seed resulted in approximately 64,250 plants/ha on south and west aspects. In contrast, approximately 6,000 plants/ha initially established on nonchained sites with similar aspects. Chaining was more beneficial on south and west slopes, where a greater amount of bare ground appeared, than on north aspects supporting a greater amount of herbaceous ground cover. Chaining not only increased seedling establishment but provided more uniform stands.

Seeding Rates

Planting big sagebrush seeds at rates between 0.11 and 0.22 kg/ha PLS is normally sufficient for broadcast and surface compaction seedings. Increasing seeding rates to 0.67 kg/ha PLS increased seedling density at the Three Creek Well and Crows Nest sites in southern Idaho, particularly when seeds were planted on the soil surface with or without compaction. Weed control measures and dry weather conditions were much more important than the amount of seed planted in regulating seedling density.

Compact seeding using the “Sagebrush Seeder” developed by Michael Boltz, Idaho BLM, resulted in uniform and high-density seedlings of Wyoming big sagebrush on a number of sites (Boltz 1994). Nearly equal numbers of sagebrush seedlings established from surface and compact seedings. Numbers exceeding 37,065-49,420 plants/ha were not uncommon. Natural thinning usually results in ultimate survival of approximately 10 to 15% of the seedlings that survive the first year.

Benefit of Snow Cover

Germination beneath snow cover is extremely beneficial to the establishment of big sagebrush seedlings. Snow cover ensures availability of soil water, maintenance of appropriate temperatures, and protection from frost. Under semiarid conditions, it is unlikely that big sagebrush seedlings will establish except in years when snow accumulates in late winter (Monsen and Meyer 1990). Collection and retention of snow on open and barren sites permits a high proportion of planted seeds to establish.

Attempts to modify soil surfaces by creating shallow depressions, deep furrows, and berms to collect snow and enhance sagebrush seed germination on large burns in Idaho and Utah have generally not been successful. Snow fences between 0.9 and 1.2 m tall have collected sufficient snow to ensure germination of big sagebrush seeds. This technique can be used in limited situations over large distances.

Broadcast seeding on snow has been a reliable technique for seeding big sagebrush on pinyon-juniper and sagebrush sites in Utah (Plummer et al. 1970). Acceptable stands have also been attained in big sagebrush steppe communities in Idaho using the same practice. Planting can be delayed until favorable amounts of snow accumulate in mid-winter. Aerial seeding of large tracts can be completed in relatively short periods.

Value of Nurse Crops and Associated Vegetation

Seeding big sagebrush on large open sites within the Intermountain region is complicated by the presence of competitive weeds and unfavorable seedbed conditions. Reestablishment of big sagebrush from natural seeding or direct seedings in areas dominated by cheatgrass has not been successful. Young and Evans (1989) recorded no recruitment of big sagebrush seedlings over a 4-year period at 5 sites dominated by cheatgrass in Nevada. Similar responses have been observed over extensive areas in the West. Weeds can be eliminated or their density reduced by mechanical tillage or application of herbicides. Wagstaff and Welch (1991) found elimination of cheatgrass by fall tillage resulted in an increase in big sagebrush seedling recruitment, due in part to an increase in seed production from existing sagebrush plants.

Replacement of cheatgrass with perennial grasses, principally crested wheatgrass (*Agropyron cristatum*), has been a common practice. However, natural recruitment of big sagebrush in stands of crested wheatgrass has been quite variable. Recruitment is site- or habitat-dependent but is based on grazing practices, presence of a seed source, and climatic conditions (Hironaka et al. 1983). Natural recruitment of sagebrush in grazed pastures has often occurred within 20 to 30 years, requiring renovation to maintain grass productivity.

Frischkencht and Bleak (1957) reported heavy seasonal grazing weakened understory crested wheatgrass and hastened big sagebrush recruitment. Although not fully confirmed, sagebrush recruitment in introduced perennial grass stands appears to occur most often in areas receiving greater than 304-356 mm of annual precipitation. Big sagebrush invasion is much slower and often is not encountered on sites receiving less rainfall. Unless shrub seedlings are able to establish at the same time seeded grasses are planted, additional recruitment is often sparse on these sites. Extensive areas of crested wheatgrass exist that remain devoid of big sagebrush, even where native stands of sagebrush exist nearby to provide a seed source.

Planting native herbaceous plants (forbs and grasses) normally favors big sagebrush recruitment. Native herbs have evolved with the native shrubs, and natural recruitment of both groups occurs following disturbances. Frischknecht and Bleak (1957) reported more big sagebrush seedlings encroached into grazed and nongrazed plots of bluebunch wheatgrass (*Pseudoroegneria spicata*) than into crested wheatgrass. There are numerous areas in southern Idaho and central Utah where big sagebrush has been able to reestablish amid an understory of native herbs. Competition with native herbs does occur, but the early-season growth habit of Sandberg bluegrass (*Poa secunda*) and other species allows for shrub seedling survival during favorable years. Replacing weeds with native understory herbs is necessary to ensure reestablishment and perpetuation of big sagebrush. Careful management of native sites to allow slow recovery of shrubs is equally critical.

Naturally recruiting and planted rubber rabbitbrush and low rabbitbrush (*Chrysothamnus viscidiflorus*) have been highly successful in providing stabilization on disturbed big sagebrush sites. These species are capable of establishing and spreading to sites occupied by cheatgrass. Once established, both rabbitbrush species facilitate recruitment of big sagebrush seedlings. Eventual colonization of mine disturbance by rubber rabbitbrush ultimately led to the establishment of big sagebrush plants within a 10-year period in northern Nevada (Meyer 1994). Similar responses occur on disturbed rangelands, although the recovery process can be much slower due to lack of an adequate seed source of big sagebrush. Planting disturbed areas to rabbitbrush to facilitate sagebrush establishment is feasible and ecologically practical. Rabbitbrush plants aid in trapping snow, moderating temperature extremes, accumulating litter, and perhaps enhancing soil microbiota that may be important for big sagebrush recruitment.

Planting Practices

Mixed Seedings

Big sagebrush is usually planted with other shrubs and herbs. Mixed plantings can be established if seeds of big sagebrush are planted in separate rows from more rapidly developing grasses. Partitioning seed boxes and drills to allow big sagebrush to be planted in separate furrows from grasses usually reduces competition to allow establishment of all species. Aerial or broadcast seeding also provides sufficient separation of seeds to ensure development of uniform stands.

Interseeding big sagebrush and associated shrubs in stands of perennial grasses, including crested wheatgrass, is a common practice (Stevens 1994). Mechanical tillage, scalping, or application of herbicides can be used to remove existing species and provide a seedbed free of competition. Guinta et al. (1975) found shrub seedling establishment in stands of perennial grasses was much higher when shrubs were seeded in strips that were at least 0.6 m wide. Van Epps and McKell (1978) found that clearings 1 m wide provided optimum spacings for shrub seedings in established stands of crested wheatgrass.

Broadcast seeding big sagebrush, followed by pipe harrowing or chaining, has been an accepted technique for seeding into stands of perennial native grasses. Sufficient reduction of herbaceous competition can be achieved to establish a uniform density of shrubs. Recovery of the perennial understory usually occurs within 1 to 3 years.

Greater acceptance and recognition of the need to restore disturbed sagebrush communities has developed in recent years. Appropriate techniques and practices are in place to restore big sagebrush communities, although treatments can be expensive and success in arid sites is uncertain.

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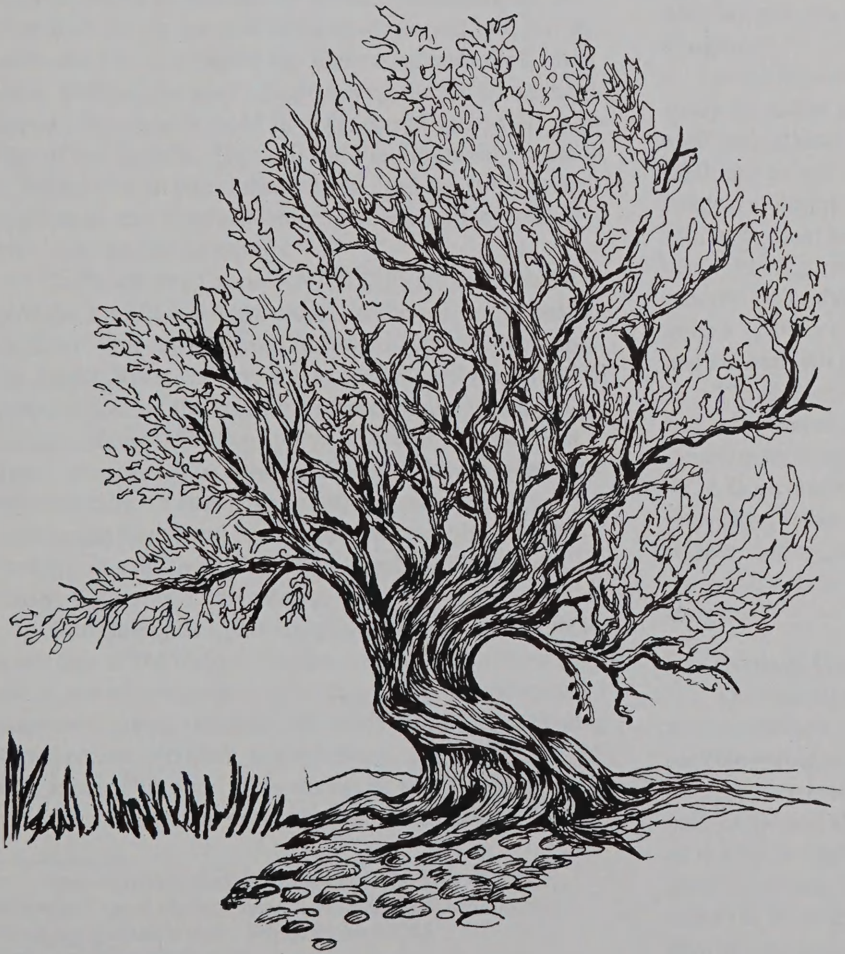
Snake River Birds of Prey National Conservation Area (NCA) Session



INTRODUCTION TO THE SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA

John Sullivan

On August 4, 1993, President Clinton signed Public Law 103-64, establishing the Snake River Birds of Prey National Conservation Area (NCA) for the purpose of "... the conservation, protection, and enhancement of raptor populations and their habitats . . . ." The enabling legislation allows existing multiple uses to continue unless they are determined to be incompatible with the purposes for which the NCA was established. Those existing uses include military training, livestock grazing, and recreation.

The NCA encompasses 485,000 acres (196,425 ha) of public land along 81 miles (130 km) of the Snake River in southwest Idaho. The river lies within a deep canyon that is surrounded by a vast upland plateau. At first glance, the plateau looks undistinguished, but it holds the key that makes this area so valuable for birds of prey. During the past 10,000 years, desert winds have deposited a deep layer of finely textured soil on the north side of the canyon. This soil and the plants that grow in it support large populations of ground squirrels and jackrabbits that supply the main food source for birds of prey, also known as raptors.

Cliffs towering up to 700 feet (213 m) above the river provide countless ledges, cracks, and crevices for nesting raptors. The combination of ideal nesting habitat in the Snake River Canyon and extraordinarily productive prey habitat on the adjacent plateau make this a place like no other for birds of prey. The area is actually a giant, natural raptor nursery. About 700 pairs of raptors representing 15 species, including golden eagles, burrowing owls, and prairie falcons, nest here each spring. In addition, nine other species use the area during their annual migrations.

More than just a gathering spot for raptors, the NCA hosts one of the nation's largest concentrations of badgers and is one of the few places in Idaho to see black-throated sparrows. Approximately 260 wildlife species, including 45 mammals, 165 birds, 8 amphibians, 16 reptiles, and 25 fish, inhabit the area. This variety of species prompted the entire NCA to be designated as a Watchable Wildlife Area. Although Dedication Point and the Snake River Canyon are the most popular areas for viewing wildlife, there are three additional recognized Watchable Wildlife sites within the NCA: the Ted Trueblood Wildlife Area, C.J. Strike Wildlife Management Area, and Bruneau Dunes State Park.

RECREATION

In addition to being a raptor-watching hot spot, the NCA provides numerous and varied recreational opportunities. Most of the visitor use is land-based, including sightseeing (nature study and archaeological-site viewing), on-trail motorized vehicle use (cars, trucks, jeeps, motorcycles, and ATVs), horseback riding, hiking, hunting and recreational shooting, mountain biking, picnicking, and camping.

From March through June, sightseeing and nature study associated with nesting raptors attract local, national, and international visitors. This time of year is also the peak use period for varmint hunters, target shooters, hikers, and mountain bikers. Water-based recreation, including float boating (rafting, kayaking, and canoeing), power boating, and fishing, are popular during the warmer months along the Snake River and on C.J. Strike Reservoir, a 7,500-acre (3,038-ha) impoundment of the Snake and Bruneau Rivers in the southeastern portion of the NCA.

The sheltered canyon areas of the NCA offer spring and fall weather conditions that average 5°-10°F warmer than temperatures in nearby Boise. Because of this, the NCA is increasingly popular with the public because it provides opportunities to recreate outdoors in the late winter, spring, and fall, when many higher-elevation recreation areas are unpopular or inaccessible due to weather.

LIVESTOCK GRAZING

The bulk of the NCA is composed of two large common allotments – the Sunnyside spring/fall allotment and the Sunnyside winter allotment. These allotments, which contain very few cross fences, are grazed in common by both cattle and sheep. Because of a lack of surface water in the NCA, the affected ranchers must haul water to their livestock. And, because many of the roads that cross the NCA are too rough for water trucks, livestock distribution and grazing intensity vary greatly throughout

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the area, with many areas receiving little if any grazing because of lack of water. BLM has begun a study to determine whether livestock grazing is achieving the objectives outlined in the Kuna Management Framework Plan, the land-use plan for the area. If objectives are not being reached, changes in livestock management will be initiated to mitigate those effects.

MILITARY TRAINING

Beginning in the 1940s, the Department of Defense began military training in what is now the NCA. The Idaho Army National Guard (IDARNG) began training in the 138,000-acre (55,890-ha) Orchard Training Area (OTA) in 1953. The OTA, which lies wholly within the NCA, consists of a 58,000-acre (23,490-ha) Impact Area into which trainees fire live artillery and small-arms rounds. The remaining 80,000 acres (32,400 ha) consist of maneuver areas, where trainees learn how to operate tanks and other tracked and wheeled vehicles. In 1979, BLM and IDARNG signed a Memorandum of Understanding (MOU) authorizing IDARNG's use of the OTA and specifying the responsibilities of each agency. The MOU was updated in 1985. No environmental documentation has been completed to address the cumulative impacts of military training. When adequate funding becomes available, IDARNG plans to initiate an environmental impact statement to address those impacts.

WILDFIRE

Wildland fire poses one of the most serious threats to the health and future of the NCA. Summer lightning storms and heavy public use make the area particularly susceptible to wildfire. Since the late 1970s, wildfires in the NCA have burned some 350,000 acres (141,750 ha). However, because many of those acres have burned more than once during that time, about 250,000 acres (101,250 ha) (a little more than half of the NCA) have actually been impacted. Replacing the shrubs and grasses has proven to be extremely difficult due to the dry desert climate.

Large-scale replacement of native shrubs and perennial grasslands by annual weeds, catalyzed by dramatic increases in the size and frequency of wildfires, is causing significant declines in important prey (black-tailed jackrabbits and Paiute ground squirrels [formerly known as Townsend's ground squirrels]) and raptor species (golden eagles and prairie falcons). Annual vegetation forms a continuous mat of fine fuel and has, over time, changed the natural fire cycle in the NCA. If present trends continue, the NCA will become completely converted to annual vegetation that will not be able to support the abundance and diversity of birds of prey the area was established to protect. In addition, neighboring communities face increasing threats from fast-moving wildfires.

The BLM will address these issues through a science-based strategy to reduce wildfire and annual weeds and restore native shrublands and perennial grasslands. The program would utilize the NCA as a demonstration project for developing and testing techniques that would have practical application in other areas across the Intermountain West. To begin this process, BLM, the Society for Ecological Restoration, U.S. Geological Survey, and Boise State University cosponsored the Sagebrush Steppe Symposium in June 1999 to provide a venue within which scientists and land managers involved in sagebrush and salt-desert shrub habitat restoration could discuss issues of mutual concern. Following the symposium, BLM sponsored a workshop involving about 60 internal and external scientists and practitioners who examined technical issues involved in habitat restoration and made science-based recommendations which will be carried forward in future planning documents and will be used to evaluate future land-use and habitat-restoration proposals.

THEN AND NOW:

Changes in Vegetation and Land Use Practices in Southwestern Idaho Sagebrush Lands, with Emphasis on Sagebrush and Former Sagebrush Lands of the Snake River Birds of Prey National Conservation Area North of the Snake River

Dana Quinney

INTRODUCTION

Once a shrub-grassland, the Snake River Birds of Prey National Conservation Area north of the Snake River was dominated, in presettlement times, by big sagebrush (*Artemisia tridentata*) and sagebrush-winterfat (*Krascheninnikovia lanata*) mosaic, subtended by perennial bunchgrasses and many species of forbs. Grazing of domestic livestock, the introduction of exotic annual plant species, and the resulting increase in fire size and frequency caused significant changes in the native vegetation prior to 1950. Between 1950 and 1994, more changes occurred, including increased shrub losses, military land use, and modernization of grazing and range management practices. Since 1994, still more changes have occurred.

PRESETTLEMENT CONDITIONS

The western Snake River Plain was a shrubland. Before European settlement, a vast sea of gray-green sagebrush occupied thousands of square miles (McArdle 1936, Piemeisel 1938, Blaisdell 1953, Ellison 1960, Vale 1975). In the Snake River Birds of Prey National Conservation Area north of the Snake River and south of Interstate 84 (hereafter called NCA North), most of the sagebrush was Wyoming big sagebrush, (*A. tridentata* ssp. *wyomingensis*), although considerable basin big sagebrush (*A. t.* ssp. *tridentata*) was also present, along with small lenses of silver sagebrush (*A. cana*) and threetip sagebrush (*A. tripartita*). Sagebrush stands composed of these species still occur in the NCA North (personal observation, 1999).

Stands of sagebrush were vast. The early European settlers wrote of "seas of wormwood" (sagebrush) stretching as far as the eye could see (Vale 1975, Yensen 1982). Sagebrush in very large stands existed in the NCA North for a hundred years after European settlement. During the early 1960s, between Boise and Mountain Home, this author recalls sagebrush as far as the eye could see south of the highway that eventually became Interstate 84.

In presettlement times, stands of sagebrush in the NCA North were open-canopy communities underpinned by native perennial grasses and forbs (Blaisdell 1953, Ellison 1960), with Thurber needlegrass (*Achnatherum thurberiana*) a dominant in the understory (Yensen et al. 1992). Great Basin wildrye (*Elymus cinereus*) was one of the most visible of these species. Pioneers noted that the sagebrush between Boise and Mountain Home appeared like "a field of wheat," with tall rye stems waving above the tops of the sagebrush (Ferrin 1935; O.R. Hicks, Idaho pioneer, personal communication). In the writings of emigrants who crossed southwestern Idaho, the most commonly mentioned native forb is balsamroot. Some noted that this group of showy yellow flowers (*Balsamorhiza*, probably including both *sagittata* and *hookeri*) was so abundant that "mile after mile" of the sagebrush land was colored yellow during spring (Vahlberry 1940; O.R. Hicks, Idaho pioneer, personal communication; C.L. Stewart, southern Idaho resident, personal communication). No doubt biological soil crust filled the interspaces with a productive and erosion-retarding cover of mosses, algae, and lichens. Although the northern sagebrush deserts are not as high in species diversity as many other ecosystems, dozens of species occurred in the sagebrush communities of the western Snake River Plain (Yensen 1982).

Of course, in presettlement times and during European settlement, fires occurred in sagebrush stands. Because the native perennial grasses and forbs withstand occasional fire rather well, it is likely that there was little soil erosion inside the burns and that they were relatively narrow, naturally reseeding via windblown sagebrush seed (Yensen 1982).

Although Native Americans of the region had obtained the horse by about 1690 (Haines 1970), the NCA North was not grazed continuously during the presettlement era. The great herds of bison (*Bison bison*) did not occur on the western Snake River Plain (Walker 1978). Mule deer (*Odocoileus hemionus*), wapiti (*Cervus elaphus*), and pronghorn (*Antilocapra americana*) wintered on the western Snake River Plain in the tens of thousands (Vahlberry 1940, Idaho State Historical Library photographs) but in spring moved to areas at higher elevations

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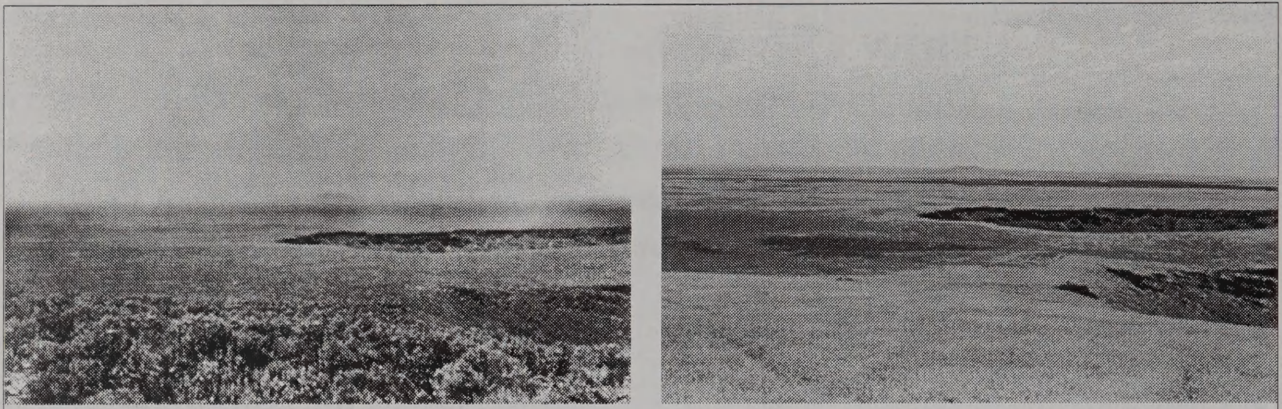


A view west from Crater Rings, a unique geologic formation in the NCA North, was taken by Israel Russell in 1901 (left) and shows a "sea" of big sagebrush. The feature on the far horizon is Cinder Cone Butte (USGS photographic collection). Photographed again in 1981 (right) by the author, the scene is now dominated by exotic grasses. The dark strip just below the far horizon is irrigated agriculture. The small patch of big sagebrush near the left center of the scene has since burned.

where water and green grass were available. Native American-owned horses wintered near the Snake River and traversed the sagebrush plains at times but were not kept in large numbers and were not pastured in the area when surface water was not available. Horses need water each day, and there was little water to be had here between June and October (Keith 1976; O.R. Hicks, Idaho pioneer, personal communication). Year-round grazers of the NCA North were ground squirrels, rabbits, and other small wildlife species (Yensen 1982).

Pioneer Settlement, Early Grazing Practices, and Vegetation Changes

Grazing of the NCA North by domestic livestock essentially began during the Oregon Trail years, from the 1840s through the 1850s. Oregon-bound pioneers grazed a corridor of the Snake River Plain so wide that their livestock had to be driven several miles from the corridor to get enough forage (Unruh 1979). Each year many of these emigrants would winter in Boise if they were so late on the trail that deep snow would preclude their crossing the Blue Mountains in Oregon. It became a common practice for emigrants wintering in Boise to turn their livestock out to graze on the "white sage plains" (winterfat stands) south of Boise, gathering up the animals and moving on in the spring when the passes of the Blue Mountains opened again (Fulton 1965).

Cattle came to the NCA North in the 1870s, when meat hunters had exhausted nearly all the big game in the Owyhee Mountains to feed the miners in Silver City (Haines 1970). Growing towns, including Boise, were also a market for beef in the 1870s. Cattle were trailed to Idaho from Texas and other southern regions and grazed in the NCA North. David Shirk, a cattleman involved in the first cattle drives to southwestern Idaho, remarked that "there was worlds of white sage (winterfat) at that time" (Shirk 1956). In the early 1870s, herds of market cattle were wintered in the NCA North. At about this time, many large herds of cattle were trailed through southwestern Idaho, both east and west (Stewart 1936, Galbraith and Anderson 1971) and, in addition, many cattle ranches were established in Ada, Elmore, and Owyhee counties (Rinehart 1932, Hanley and Lucia 1973, Sharp and Sanders 1978). These were not cow-calf operations; they included large numbers of steers. Many of these cattle were not harvested; herds were allowed to increase on the range to serve as an increase in "capital" for the ranchers, as bankers loaned money based upon livestock numbers (Stewart 1936). This practice kept many animals on the open range year-round.

Sheep were also trailed in, first from California in the 1860s and then from Utah and Nevada (Wentworth 1948). After 1870, numbers of range sheep in Idaho grew rapidly and wool became the industry's most important product. Sheep ranchers owned tens of thousands of animals each. These herds also grazed the NCA North during the winter and early spring months (Wentworth 1948). By the early 1870s, large trail drives of sheep were moving each summer through southwestern Idaho, eating grasses, forbs, winterfat, and bud sagebrush, then returning to home ranches as far away as Nevada, Oregon, and Utah. The 1880s were boom times for the Idaho sheep industry (Wentworth 1948, Yensen 1982).

Because there was no water hauling in the 19th century, cattle and sheep were driven to higher pastures in the foothills in late spring and then into the forests as grasses cured and water holes dried up. With the coming of fall, livestock would return to the lower desert ranges to feed upon the dried perennial grasses called by stockmen "spontaneous hay" or "standing hay" (Hodgeson 1948, Nettleton 1978).

Horses, the work engines of this era, were also grazed in the NCA North. Local ranchers grazed thousands of horses in the NCA North in the winter, when many of them were not needed for agricultural work. A few local ranches maintained as many as 2,000 horses each (Nettleton 1978). Many horses were turned out to forage on the open range in winter and became the nucleus of wild horse herds that roamed the area beginning in the 1870s. These wild horses, of course, were not herded to greener pastures in the spring but remained in the desert all year, damaging riparian areas and young seedlings of desert plants (McKnight 1964).

Competition for forage between cattle and sheep (and horses) began, and overgrazing of the range was noted as early as 1880 (Wentworth 1948). Intense competition between cattle ranchers and sheepmen for livestock forage had serious consequences for the native vegetation. When sagebrush-grass habitat is intensively grazed, native perennial grasses are eliminated and sagebrush tends to form dense, monotypic stands (Blaisdell 1949). Winterfat plants grazed to 30% of the above-ground volume require 10 or more years of rest to regain their original size; without rest, they may die and eventually be replaced by shadscale (*Atriplex confertifolia*) or other less nutritious plants (Cox 1977). Winterfat was very heavily grazed at this time. By 1890, the native perennial grasses (the larger bunchgrasses like bluebunch wheatgrass [*Pseudoroegneria spicata*], Indian ricegrass [*Achnatherum hymenoides*], Great Basin wildrye, and needlegrasses [*Achnatherum thurberianum* and *Hesperostipa comata*]) were, for all practical purposes, no longer present on southern Idaho ranges (Hodgeson 1948). Many very palatable forbs cannot withstand even light grazing and may be eliminated from large areas. Sheep tend to select forbs when palatable ones are available (Griffiths 1902). Native forb species began to disappear from thousands of square miles of the Snake River Plain (Vahlberry 1940). As can be seen by pedestalling of shrubs and grasses in many historic photos (Idaho State Historical Society photo collection) taken around the turn of the past century, topsoil loss also had begun to occur.

Monotypic stands of sagebrush resulted from elimination of many grasses and forbs, and stockmen set range fires to get rid of the sagebrush. This practice continued for decades (Griffiths 1902; Pechanec et al. 1937; Young et al. 1979; O.R. Hicks, Idaho Pioneer, personal communication).

The semiarid climate of the western Snake River Plain has a history of extreme variation in amount and timing of precipitation (Wernstedt 1960), and therefore it is difficult for damaged vegetation to recover quickly. With desert grasses and forbs already seriously depleted and cattle and sheep numbers continuing to increase, the harsh winter of 1889-1890 was a disaster for the livestock industry in southwestern Idaho. Tens of thousands of cattle, sheep, and horses died on the range that winter on the western Snake River Plain (Wentworth 1948, Yensen 1982).

ALIEN INVASIONS

By 1900, there were significant voids in the understory vegetation (Hodgeson 1948, Sharp and Sanders 1978). The damaged ranges of southwestern Idaho were ripe for takeover by aggressive, fast-growing plant species. Russian thistle (*Salsola kali*) was likely the first invader, spreading first through seed dispersal via irrigation canals. Various exotic mustards (*i.e., Sisymbrium altissimum, Lepidium perfoliatum*) soon followed (Weaver 1917); and then appeared the European winter annual, downy brome, hereafter referred to by its most common name, cheatgrass (*Bromus tectorum*).

Many stockmen were very enthusiastic about the new "wonder grass" and at first believed it to be superior to the natives that had been decimated (Leopold 1941). Their enthusiasm was to be short-lived. Cheatgrass is very flammable. As early as the 1940s, it was recognized that five times more fire crews were needed to stand by on cheatgrass ranges than on any other (Stewart and Hull 1949). Cheatgrass seeds withstand fire well, and the species increases rapidly when fire is combined with grazing (Stewart and Young 1939, Leopold 1941, Ellison 1960); cheatgrass-dominated acreage in southwestern Idaho expanded.

By 1930, cheatgrass was widely distributed in southern Idaho (Rinehart 1932). By 1949, cheatgrass dominated 4 million acres (1.6 million ha) of Idaho rangeland (Stewart and Hull 1949), and its dominance has continued to increase. The burn-reburn cycle in this region was altered by cheatgrass and other exotic annuals, and wildfires occurred with increasing frequency and severity (Wright and Klemmedson 1965).

DROUGHT, RECOVERY, AND MODERN LAND USE PRACTICES

During years when grain harvests in Europe were disrupted by World War I, many homesteaders filed Desert Land Entry claims in the southwestern Idaho desert and cleared hundreds of thousands of acres of land to grow crops. Then came the long dry cycle of 1914-1934, causing many of these farms to fail. Cheatgrass invaded these wastelands and spread out into the nearby range at a time when the livestock industry had just been through a period of expansion following the disastrous winter of 1889-1890 (Weaver et al. 1935, Stewart 1936, Pechanec et al. 1937, Talbot and Cronemiller 1961). Another livestock crash resulted, bottoming out in 1934 and resulting in the Taylor Grazing Act, which introduced governmental range management to the public lands of the region (Young et al. 1979, Yensen 1982).

Range improvement practices began, continuing up to the present, including reseeding ranges to perennial grasses, controlling livestock movement by means of fences, standardizing grazing allotments by permit, and other practices. Carrying capacity has improved considerably since 1934 (Young et al. 1979). However, considering that range standards were delineated in the 1930s, at a time when the general range condition was extremely poor, it is not surprising to see improvement (Young et al. 1979).

After the 1950s, sheep numbers declined while cattle numbers increased, and ewe-lamb and cow-calf operations are now the rule. Steers and wethers no longer graze the range. The practice of hauling livestock water in the NCA North and other semiarid areas, begun in the 1950s, has allowed longer and more intensive use of dry southwestern Idaho ranges in spring, early summer, and fall (Yensen 1982).

At some point, the NCA North (and some adjacent public land) was divided into two large, multiple-permittee grazing allotments for sheep and cattle: Sunnyside spring-fall (mostly current and former big sagebrush and big sagebrush-winterfat mosaic habitats) and Sunnyside winter (current and former salt-desert shrub habitats) (Yensen 1982).

More exotic plant species have invaded the area, including halogeton (*Halogeton glomeratus*), bur-buttercup (*Ranunculus testiculatus*), and medusahead (*Taeniatherum caput-medusae*) (Yensen 1982).

In 1953, the Idaho Army National Guard (IDARNG) began training in part of the NCA North. In the early 1980s, the Orchard Training Area's boundary was redrawn to move military training farther from the Snake River Canyon. IDARNG began an environmental management program in 1987; and, in 1988, they implemented a quick-response policy of fighting all fires – whether or not they occurred in the Impact Area. In 1990, staging of maneuvers and bivouacking in sagebrush stands were prohibited; the following year, 700 acres (284 ha) of big sagebrush habitat were placed off limits to military training to protect rare plant species (personal observation).

Between 1953 and 1992, livestock were grazed inside the 56,000-acre (22,680-ha) Impact Area, but as a rule, livestock watering tanks were not allowed more than a few meters inside the perimeter road (Range Road). In 1992, this policy changed, and livestock watering tanks were allowed inside the Impact Area, everywhere except the 2,200-acre (890-ha) core, which was fenced. In addition, since 1992, firing on the ranges has been greatly reduced for a 45-day "window" during the spring grazing period to allow more extensive use of the Impact Area by livestock (personal observation).

Fire, Before 1994

With increased cheatgrass dominance came increased fire size and frequency. The more big sagebrush habitat burns, the greater the danger of burning adjacent big sagebrush habitat (Wright and Klemmedson 1965); thus, fire size and frequency has continued to increase, punctuated by breaks in the trend due to low-rainfall years with low fuel production. The burn-reburn cycle is now so short that reestablishment of native vegetation after a burn has become unlikely (Whisenant 1990). Thus, many burned areas have become more or less permanent stands of exotic annuals, and often, cheatgrass monocultures (Young and Evans 1973, Yensen 1982, Whisenant 1990).

The practice of planting new burns with perennial grasses to improve forage, prevent cheatgrass domination, and reduce soil erosion evolved during the 20th century. Beginning in the 1930s, controlled burns were conducted by ranchers and agencies to remove sagebrush and weeds in order to plant exotic perennial grasses and improve livestock access to existing native grasses (Pechanec et al. 1954, Vale 1974, Yensen 1982).

Some burns and rehabilitative seedings had occurred by 1979, when the first Snake River Birds of Prey Area boundaries were being refined. Prior to that, most of the NCA North was dominated by native shrub-grassland (USDI 1979). In the early and mid-1980s, several large fires tipped the balance in favor of non-shrub habitats. Those fires included the Coyote Butte fire of 1981 and the Black Butte fire of 1985, each burning more than 40,000 acres (16,200 ha) within the NCA North, much of which was sagebrush. In the late 1980s, "greenstrip" practices (planting wide strips of perennials to serve as firebreaks) were developed and implemented in the NCA North (Kochert and Pellant 1986).

Post-1994 Significant Changes in Vegetation and Land Use practices

We tend to think of vegetation history as something in the distant past, but sweeping changes have occurred in the NCA North since 1994. Much of the remaining sagebrush-winterfat and Wyoming big sagebrush habitat outside the Orchard Training Area burned between 1994 and 1997. No longer shrub-grassland, the lands now consist mostly of exotic grasses, especially cheatgrass. Other areas have been seeded to stands of introduced perennials, including crested wheatgrass (*Agropyron* spp.) and Russian wildrye (*Elymus junceus*). Shrub species have been included in many seedings; however, at present most of those shrubs are immature, so the seedings are currently grassland habitat.

This recent loss of winterfat-big sagebrush mosaic and big sagebrush stands near the Snake River Canyon and east of Swan Falls Road was a significant habitat change. The fires of the 1990s removed shrubs from tens of thousands of acres in the areas where radioed prairie falcons were most frequently logged in a study completed just before the fires (Marzluff et al. 1997) and removed black-tailed jackrabbit (*Lepus californicus*) habitat from lands adjacent to Snake River Canyon cliffs where golden eagles (*Aquila chrysaetos*) nest. Effects of this habitat change on populations of vertebrates seem likely.

Some stands of virtually cheatgrass-free big sagebrush still exist in the NCA North. Much of the remaining big sagebrush habitat is in the Orchard Training Area. Since 1988, the Orchard Training Area has lost approximately 1% of its big sagebrush (IDARNG GIS and vegetation plot data, 1998). Much of this remaining sagebrush is in 1 stand of approximately 22 mi^2 (57 km^2), having an understory of native grasses (IDARNG unpublished vegetation plot data, 1998).

Military use of the Orchard Training Area has changed in recent years as well. In 1996, in response to recommendations made in the BLM/IDARNG Research Project Final Report (USDI 1996), IDARNG severely restricted off-road vehicle maneuvers in sagebrush stands. In 1998, except for a handful of low-level weekend exercises, there was no maneuver training in the Orchard Training Area. Instead, the 1998 Training event took place at Fort Irwin, California (IDARNG Range Control scheduling database; personal observation).

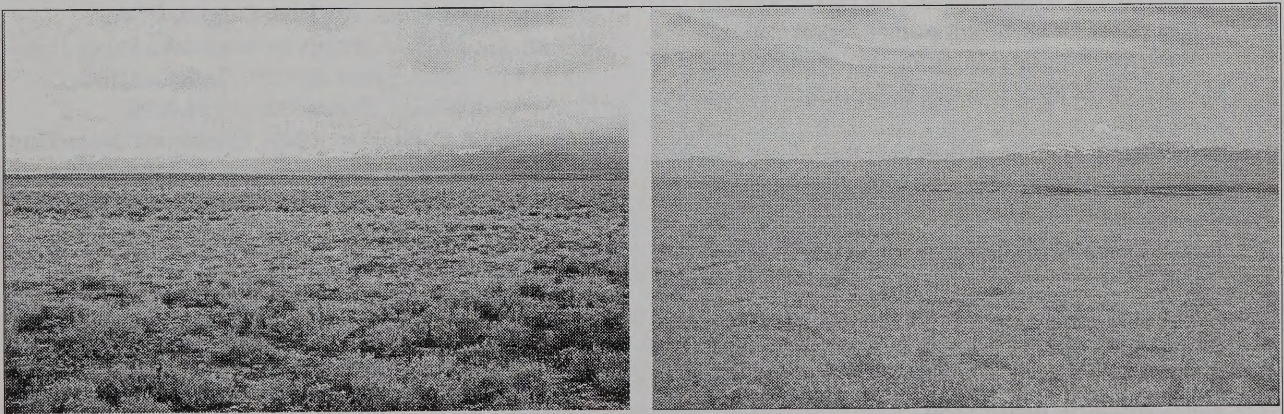
Discussion and Conclusions

The former big sagebrush-dominated area in the NCA North now hosts several habitats: a) stands of weeds dominated by cheatgrass; b) stands of weeds dominated by exotic mustards and other annuals; c) stands of lower-seral native grasses; d) stands seeded to exotic perennials; and e) stands of depauperate winterfat, winterfat-big sagebrush, and big sagebrush communities, with understories dominated for the most part by lower-seral native grasses, essentially lacking a native forb component (IDARNG vegetation plot data, 1998).

The following are personal observations: In the NCA North, some plant species that were formerly common in the understory of big sagebrush communities are locally extirpated or nearly so. It is difficult, for example, to find yellowbells (*Fritillaria*) in the NCA North. Desert Indian paintbrush (*Castilleja chromosa*) now exists at only a few locations. In 20 years of field work in the NCA North I have never seen a plant of ball-head waterleaf (*Hydrophyllum capitatum*), bluebells (*Mertensia*), or violet (*Viola*). Arrowleaf balsamroot, a once-abundant forb valuable both to livestock and to wildlife, has virtually disappeared during the past 20 years. These are all taxa commonly found in southwestern Idaho in comparable big sagebrush communities. Many forbs, such as pussytoes (*Antennaria dimorpha*), tapertip hawksbeard (*Crepis acuminata*), white forget-me-not (*Cryptantha*), and biscuitroot (*Lomatium*), which were at one time very common or abundant in southwestern Idaho sagebrush communities, have nearly vanished from the NCA North. Most of the native bunchgrasses in the NCA North are of two species, Sandberg bluegrass (*Poa secunda*) and bottlebrush squirreltail (*Elymus elymoides*), which are early seral, relatively small, short-lived species (Welsh et al. 1987; IDARNG unpublished vegetation plot data, 1998).

On a more optimistic note, there are still big sagebrush stands in the Orchard Training Area with an understory of the late seral species, Thurber needlegrass. These communities also have populations of the rare species slick-spot peppergrass (*Lepidium papilliferum*) and woven-spore lichen (*Texosporium sancti-jacobi*) in the understory (IDARNG unpublished vegetation plot and rare plant data, 1998).

Additional personal observations: In the past five years, many old burns formerly dominated by native perennial bunchgrasses have been taken over by cheatgrass. Several "new" exotics are essentially massing forces on the edges of the NCA North, with the potential to invade the remaining native plant communities. In my opinion, the most serious of these threats is that posed by rush skeletonweed (*Chondrilla juncea*), a perennial composite. Each plant typically produces hundreds to thousands of seeds per year (Whitson 1996). Rush skeletonweed began appearing along Simco Road (Elmore County, near the boundaries of the NCA North) in 1990



The image on the left (1988) and the one on the right (1999) illustrate shrub habitat loss in the NCA North in recent years. This area, east of the Snake River Canyon within 2 miles (3.2 km) of Swan Falls Dam, was formerly dominated by winterfat and winterfat-big sagebrush mosaic and is now a stand of cheatgrass (author photos).

and by 1997 was appearing around livestock-watering sites within the NCA North (personal field notes, 1990-1998).

This author believes it is time to search the NCA North for remnant individuals of plant species on the verge of local extirpation, collect their seed, and replant them into increaser fields and into selected sagebrush stands that are protected from fire, military use, and livestock grazing. Rapid-response fire protection is also necessary if any native sagebrush communities are to survive. And finally, it may be time to take active measures to prevent introductions of more exotics into the NCA North. This could be done by requiring that military vehicles be washed before entering the Orchard Training Area and that livestock be cleaned and held in corrals for several days (until weed seeds are eliminated from alimentary tracts) before the animals are allowed to enter the NCA. It is time to direct resources toward preserving and restoring the native plant communities of the NCA North.

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EFFECTS OF DISTURBANCE ON SHRUB STEPPE HABITATS AND RAPTOR PREY IN THE SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA, IDAHO

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We examined the effects of military training and wildfires on shrub steppe habitats and 2 primary raptor prey, Paiute ground squirrels (*Spermophilus mollis*) (formerly Townsend's ground squirrels [*Spermophilus townsendii*]) and black-tailed jackrabbits (*Lepus californicus*) in the Snake River Birds of Prey National Conservation Area (NCA) in southwestern Idaho. Habitat change is a significant management concern because big sagebrush (*Artemisia tridentata*), winterfat (*Krascheninnikovia lanata*), and shadscale (*Atriplex confertifolia*) communities are rapidly being converted to large expanses dominated by cheatgrass (*Bromus tectorum*), an exotic annual grass. Since 1979, over 50% of approximately 100,000 ha (247,000 acres) of shrublands in the NCA has been destroyed and the total grassland cover has increased from 17 to 53% (USDI 1996). Habitat conversion from shrublands containing a perennial grass understory to grasslands dominated by exotic annuals has resulted in a decreased interval between repeat fires in the NCA. From 1950 to 1979, the interval between recurring fires averaged 80.5 years, compared to 27.5 years for the period from 1980 to 1994.

Combined disturbances from wildfires, military training, and livestock grazing had an additive or synergistic effect on the landscape. Regions of multiple disturbance factors experienced the greatest change in land cover, primarily loss of shrub cover, between 1979 and 1992 (USDI 1996). Separately, wildfires and military training each influenced habitats differently (Knick and Rotenberry 1997). At local scales, tracked areas in which the military trained had more bare ground and greater cover of litter and exotic annual vegetation compared to untracked sites (Watts 1998). Compared with unburned sites, habitats at burned sites had significantly less cover of lichens, mosses, total (lichen+moss) cryptobiotic crusts, shrubs, and vegetation and had significantly more bare ground and greater cover of exotic annuals and vegetative litter. At larger spatial resolutions, military tracking was associated with greater fragmentation and smaller, more closely spaced shrubland patches compared to burned or unburned regions in the NCA (Knick and Rotenberry 1997). Regions in which repeated fires had burned, such as the Range Road Interior portion of the Orchard Training Area, contained few shrublands. Using computer simulations, we estimated that complete recovery of shrublands by natural processes was not possible within a century in some burned regions of the NCA because of loss of seed sources (USDI 1996).

Conversion from shrubland and perennial vegetation to habitats dominated by annual vegetation primarily influenced populations of ground squirrels through an increased susceptibility to environmental fluctuation (Van Horne et al. 1997). During a drought in 1992, squirrel populations in habitats consisting of annual vegetation experienced greater population fluctuations, lower birth rates, and lower juvenile and adult survival compared to populations in habitats having a more drought-resistant component of perennial shrub vegetation. Grasslands dominated by exotic annual vegetation supported high population densities of ground squirrels during nondrought years. However, populations of ground squirrels were less viable in annual grasslands than in habitats with a perennial shrub component because of greater population fluctuations. Although military tracking changed vegetative cover, short-term (2-yr.) or long-term (approx. 50-yr.) effects on ground squirrel densities or behavior were not detected (Van Horne and Sharpe 1998).

Densities of black-tailed jackrabbits have declined in the NCA over 3 successive peaks in population (1971, 1979-1981, and 1990-1992) (USDI 1996). Habitat selection by jackrabbits was determined from night spotlight surveys and GIS analysis (Knick and Dyer 1997). Jackrabbits were primarily associated with large

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shrubland patches throughout the NCA and absent from highly fragmented landscapes or those dominated by grasslands. Therefore, lower population densities of jackrabbits may be associated with the large-scale loss of shrublands in the NCA. Distribution and abundance of jackrabbits in the NCA will be related to the restoration (or loss) of shrublands in the NCA.

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EFFECTS OF WILDFIRES AND MILITARY TRAINING ON RAPTORS

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Long-term studies of raptors in the Snake River Birds of Prey National Conservation Area (NCA) have shown that wildfires and other habitat alteration affect raptor species in different ways. Responses of raptors to habitat changes caused by wildfire also vary among pairs and individuals within species. In this paper, we report on how populations of 6 species have responded to large-scale losses of shrub habitat in the NCA.

We studied effects of fire on golden eagle (*Aquila chrysaetos*) habitat use, territory occupancy, and reproductive success in the NCA because golden eagles nesting in the NCA depend primarily on black-tailed jackrabbits (*Lepus californicus*) and jackrabbits in turn depend on shrub habitat. Radio-tagged golden eagles tended to avoid burned habitat, and home range sizes correlated positively ($r = 0.67$, n = 9, $P = 0.5$) with percent of area burned in the home ranges. Eagle success (percentage of pairs that raised young) at burned territories declined after major fires (Kochert et al. 1999). Pairs in burned areas that could expand into adjacent vacant territories were as successful as pairs in unburned territories and more successful than pairs in burned territories that could not expand. Success at extensively burned territories was lowest 4-6 years after burning but increased 4-5 years later. The incidence and extent of fires did not help predict which territories would have low occupancy and success rates in postburn years. Responses to fire were variable and influenced by at least 3 factors: (1) whether the nearest neighboring territory was vacant; (2) the ability to use alternative foraging habitat (i.e., farmland, cliff, talus, riparian); and (3) the underlying quality of the pair or territory (Kochert et al. 1999). The presence of a vacant neighboring territory and the amount of agriculture and proportion of shrubs within 3 km of the nesting centroid best predicted probability of territory occupancy. Nesting success during preburn years best predicted the probability of a territory being successful in postburn years. Burned territories with high success rates during preburn years continued to have high success rates during postburn years, and those with low success in preburn years continued to be less successful after burning (Kochert et al. 1999). A significant decline in the number of nesting golden eagle pairs between 1971 and 1994 and the general decline in black-tailed jackrabbits suggest a possible reduced carrying capacity for golden eagles in the NCA as a result of shrub loss (Steenhof et al. 1997).

Fire apparently had little or no effect on 4 species of raptors that nest in the benchlands above the NCA canyon. Mean number of ferruginous hawk (*Buteo regalis*) pairs and mean success did not change after major fires. Nearly half of territories occupied after major fires contained >40% burned habitat within 1.5 km of the nest (Lehman et al. 1996*a*). Successful territories contained more grass habitat within 1.5 km of the nest than unsuccessful territories. Most burrowing owl (*Athene cunicularia*), northern harrier (*Circus cyaneus*), and short-eared owl (*Asio flammeus*) nests located between 1992 and 1994 occurred in burned or grassland areas (Lehman et al. 1996*b*). Burrowing owls and short-eared owls used burned and grass habitats in proportion to availability. Observations suggest that burrowing owls are now more abundant in the NCA than before widespread wildfires occurred in the early 1980s. In the early 1970s, the burrowing owl was an uncommon nesting raptor in the NCA; in 1994, we found 87 occupied territories in a 160,541-ha area of the NCA.

The relationship between prairie falcons and habitat alteration is more complex than for other raptor species. Unlike eagles, prairie falcons range up to 25 km from the canyon to feed on ground squirrels. The number of

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prairie falcon pairs found on long-term survey segments declined significantly from 1976-1997 (Steenhof et al. 1999). Early declines were most severe at the eastern end of the NCA, where fires and agriculture have changed native shrub steppe habitat. More recent declines occurred in the portion of canyon near the Orchard Training Area (OTA), where the Idaho Army National Guard conducts artillery firing and tank maneuvers. Overall prairie falcon reproductive rates were tied closely to annual indexes of ground squirrel abundance. Most reproductive parameters showed no significant trends over time, but during the 1990s, nesting success and productivity were lower in the stretch of canyon near the OTA than in adjacent areas. Extensive shrub loss by itself did not explain the pattern of declines in abundance and reproduction that we observed. Recent military training activities likely have interacted with fire and livestock grazing to create less-than-favorable foraging opportunities for prairie falcons in a large part of the NCA, but we do not fully understand the processes involved (Steenhof et al. 1999).

Managers face challenges in their attempts to regulate land uses to protect and enhance raptor populations. To provide habitat for raptor diversity, managers should continue and expand their programs to suppress wildfires and restore native shrubs and perennial grasses. We recommend that BLM design a comprehensive adaptive management program in which experimental management actions are monitored at multiple scales to determine if restrictions and restorations are achieving desired results. Managers should consider that fire affects various raptor species differently and responses vary greatly among pairs and individuals within species. Population responses of long-lived raptors to habitat alterations, including restoration, may take decades, requiring long-term population monitoring and well-designed adaptive experiments with large spatial and long-term temporal scales.

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WILDFIRE SUPPRESSION IN THE SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA

Bill Casey

ABSTRACT

Since the late 1970s, wildfires in the NCA have burned some 350,000 acres (141,750 ha). However, because many of those acres have burned more than once during that time, about 250,000 acres (101,250 ha) (a little more than half of the NCA) have actually been impacted. A number of factors have contributed to the large number of acres burned.

Since the early 1980s, frequent fires have been a major contributor in the decline of native vegetative communities and have contributed to the expansion of annual grasses, which have, in turn, significantly increased fire occurrence in the NCA The most common of these exotic annuals is cheatgrass, a highly flammable, prolific producer in wet years. Cheatgrass is easily ignited and fire spreads very rapidly. These fires typically require the use of multiple suppression forces to control them.

Weather cycles and patterns within the NCA further complicate fire suppression. Wet springs that ensure abundant annual vegetation are typically followed by hot, dry summers with frequent frontal passages that bring dry lightning and high winds. Frequent multiple fire events, with fires scattered throughout southwest Idaho, are associated with these storms.

Suppression forces are dispatched aggressively to wildfires in the NCA. Availability of firefighting resources is dependent upon the overall district fire situation and the fire's proximity to suppression resources, which are situated in several locations adjacent to the NCA. Because of these and other factors, response times can vary from a few minutes to more than an hour, depending upon the fire location. During multiple fire events, the district often has insufficient resources to contain wildfires within fire-size objectives established by resource managers.

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FUELS MANAGEMENT IN THE SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA

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INTRODUCTION

The current frequency, severity, and distribution of wildfires on the Snake River Plain are significantly higher than historical levels, especially in the Snake River Birds of Prey National Conservation Area (NCA) where nearly half (250,000 acres [101,250 ha]) of the public land burned in the 15-year period between 1980 and 1994 (USDI 1995). (Total fire acreage is closer to 350,000 [141,750 ha], as many areas burned more than once during that time.) Historically, sagebrush steppe vegetation in the Great Basin was impacted by wildfires at return intervals of 32 to 70 years (Wright et al. 1979). Today, areas dominated by cheatgrass (*Bromus tectorum*) tend to reburn at intervals of less than 5 years in parts of southwestern Idaho (Pellant 1990, Whisenant 1990). This change in wildfire frequency was initiated by the introduction of domestic livestock in the late 1800s, which resulted in widespread overgrazing of native vegetation (Yensen 1982). Native herbaceous plants were weakened or removed, allowing alien annual grasses to rapidly invade and dominate degraded rangelands (Young et al. 1972, Young and Evans 1978). These factors have led to serious resource concerns about maintenance of critical habitat for prey populations and raptors in the NCA (Kochert and Pellant 1986, USDI 1995).

Cheatgrass and medusahead wildrye (*Taeniatherum caput-medusae*) are the dominant flammable alien grasses in southern Idaho (Stewart and Hull 1949, Torell et al. 1961). In the NCA, cheatgrass is much more common than medusahead wildrye, although medusahead is expanding its range. Cheatgrass matures earlier than native perennial grasses and is easily ignited, thereby increasing the likelihood of repeated wildfires (Young et al. 1987). Fire suppression on cheatgrass rangelands is difficult due to the wide fire front and rapid rate of fire spread.

Fuels management is the manipulation of plants and litter to reduce the frequency, rate of spread, and size of wildland fire (USDI 1998). Fuels management differs from fire management in that fuels management is a proactive approach to reducing wildfires, while fire management is a reactive process to stop wildfires once they have started. Fuels management was discussed in sections of the 1995 NCA Management Plan regarding the greenstripping program and livestock use (USDI 1995).

This paper discusses fuels management practices that are currently in use or potentially available to reduce wildfire spread in the NCA. In many instances, a combination of these treatments will better reduce the rate of spread and extent of wildfire compared to a single treatment. Fuels management should not be considered the sole solution to eliminating the wildfire problem in the NCA. Rather, an integrated program of fuels management, fire suppression, public education to reduce human-caused fires, and rehabilitation after wildfire are all part of the solution.

PRINCIPLES OF FUELS MANAGEMENT

Fuels management on rangelands is directed toward modifying fuel properties to reduce extreme fire behavior. Standing dead material, litter, and live plants constitute the bulk of rangeland fuels. Fuel is the only element of the fire behavior triangle (fuel, weather, topography) that can be influenced by management actions, as neither weather nor topography can be manipulated.

Fuel Availability and Continuity

Fuel potential for combustion depends upon several factors, including the proportion of fuel that is dead, fuel particle size, moisture content, and continuity (Anderson and Brown 1988). The likelihood of a fire start and rate of fire spread increases as fuel availability and continuity increase, as is often the case with rangelands infested with cheatgrass or medusahead wildrye. These exotic annual grasses are more prone to ignition and fire spread than the native perennial and annual species, due to higher proportion of available, contiguous fuels. Early maturation of cheatgrass and medusahead wildrye, compared to native herbaceous species, increases the length of time that fuel is available and, thus, the duration of the fire season.

The effectiveness of a fuels modification project in reducing wildfire spread may be increased by implementing the following actions (singly or in combination):

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1. Disrupt fuel continuity. Fuel continuity can be disrupted by removing all or most of the vegetation (e.g., by mowing, disking, burning, or applying herbicide) or replacing cheatgrass, which grows in a mat-like pattern, with caespitose grasses (bunchgrasses), which have larger spaces between individual plants. This treatment reduces the spread of surface fires, since discontinuous fuels do not carry a fire as well as continuous fuels (Anderson and Brown 1988).

2. Reduce fuel accumulations and/or volatility. A high density of woody plants (e.g., shrubs) generates longer flame lengths compared to herbaceous vegetation and increases the probability of fire-spotting in rangelands (Schmidt and Wakimoto 1988). Big sagebrush (*Artemisia tridentata*) has a high volatile oil content (Kelsey 1986), further increasing fire severity in shrublands.

3. Increase the proportion of plants with a higher moisture content. The moisture content of the various species in the plant community governs the length of time during the fire season when fuels and fire behavior are hazardous and ignition potential is high (Anderson and Brown 1988). Increasing the proportion of plants with high moisture and low volatile oil content can reduce both the potential for ignition and the rate of fire spread.

FUELS MANAGEMENT ALTERNATIVES

There are several fuels management alternatives currently in use or potentially available for use in the NCA. Combinations of the described techniques are not included in this discussion; however, 2 or more treatments applied together may provide better fire control than single treatments. Although this paper contains an overview of techniques commonly used for fuels management, it is not intended to be an exhaustive discussion of all possible techniques.

Mowing

Fuel reduction by mowing is expensive, time consuming, and requires annual treatment. Fuels are reduced but not removed; therefore, the threat of wildfire spread is not eliminated. If mowing is done too early in the growing season, plants may regrow and remain a fire hazard. Vigor and persistence of native or seeded plants may be reduced, opening the treated area to further expansion of exotic annual grasses and noxious weeds.

Other limitations regarding the use of mechanical mowers include safety issues related to operation on slopes, rough topography, and hidden rocks. The potential for inadvertent fire starts during the mowing process is a concern in rocky areas, especially after the vegetation is dry. Mowing has little application in the NCA except along roads, in recreation sites, and in rural/wildland interface areas.

Thinning

Thinning is a fuels management option in decadent and/or dense big sagebrush stands. Increasing the distance between sagebrush plants by hand or through mechanical thinning would reduce fuel loads and continuity, thereby reducing fire spread. Schmidt and Wakimoto (1988) recommended thinning shrubs to a minimum distance of 10 feet (3 m) between plants to reduce the probability of fires spreading laterally. The distance between shrubs and the landscape configuration of the treatment would have to be refined to maintain quality prey habitats in the NCA.

Hand thinning sagebrush is cost-prohibitive on project-size treatments. A mechanical thinning tool that could meet this objective is the disk chain. With the proper configuration, this implement, which was described by Pellant (1988), could remove approximately 50% of the sagebrush and distribute seed for herbaceous species during the thinning operation. Seeding fire-resistant vegetation in the understory of thinned sagebrush stands has the added benefit of further reducing cheatgrass potential and wildfire spread. These thinned sagebrush stands normally would produce greater quantities of viable seed than the original decadent or dead sagebrush stands, resulting in recruitment of new sagebrush plants in the treated area. Thinned sagebrush stands also could serve as good collection sites for local sagebrush seed for restoration projects.

Plowing or Disking

Mechanical fuel breaks are costly to maintain and require annual treatment. They are not effective in rocky areas, are visually obtrusive, can increase erosion, and can facilitate the spread of undesirable weeds. Because all fuel is removed, mechanical fuel breaks are more effective than mowed fuel breaks in reducing wildfire spread. In general, the probability of breach increases with elevated fire intensity and presence of woody vegetation and decreases with greater fuel-break width (Wilson 1988). Because the width of most mechanical fuel breaks is generally less than 50 feet (15.2 m), the chance of breach during extreme fire weather conditions is high.

Mechanical fuel breaks are maintained along a few highways, within the National Guard training area, and around cropland within the NCA. An expansion of the current fuel-break network is not expected, given the adverse impacts described above and the values and public scrutiny associated with this area.

Livestock Grazing

The use of livestock to reduce fuel loads in cheatgrass rangelands is not a new concept. Stewart and Hull (1949) found that heavy grass use by sheep in early spring greatly reduced cheatgrass density and height. More recently, Vallentine and Stevens (1994) reviewed the literature on the use of livestock to control cheatgrass and concluded that, with appropriate management considerations (season of use, careful livestock management, and appropriate livestock forage utilization levels), cheatgrass production could be reduced. Davison (1996) encouraged the use of livestock to reduce wildfire danger with the caveat that it requires more intensive livestock management to achieve fuel-reduction objectives.

There are many factors, including season of use and size of the pasture or grazing allotment, that must be considered before using livestock to manage fuels. The Sunnyside winter allotment is one of the larger grazing units (roughly 192,000 acres [77,760 ha]) in the NCA. Livestock are in the allotment from mid-December until the end of February each year. This season of livestock use is not conducive to reducing fuels for the following fire season, since spring cheatgrass growth is the primary factor that determines fuel loads.

Most of the grazing allotments in the NCA are spring/fall season of use, with cattle being the predominant class of livestock. Spring grazing is more conducive to controlling cheatgrass than is winter grazing (Vallentine and Stevens 1994). Therefore, there is potential for using livestock to reduce fuel loads on grazing allotments in the NCA. However, heavy spring livestock use can be detrimental to other resource values.

Livestock are just 1 factor that influence fuel loads on cheatgrass-dominated rangelands. Forage production of cheatgrass, and thus fuel loads, can vary tremendously, depending on climatic conditions, particularly amount and timing of precipitation (Stewart and Hull 1949). The length of time that cheatgrass is palatable to livestock in the spring also varies considerably on an annual basis. Adjusting livestock numbers upwards to fully utilize cheatgrass in high precipitation years and totally destocking in drought years is not economically feasible for many livestock operators. In order to adequately control cheatgrass and reduce fuels sufficiently to reduce wildfires, livestock use levels may negatively affect other resource values (e.g., vigor of remnant native plants, soil stability, biological soil crust). Other invasive or noxious weeds may increase due to disturbance associated with the intensity of livestock use required to accomplish fuels management objectives.

Perhaps the best strategy to manage fuels using livestock is to concentrate animals (probably sheep, since they can be herded) to utilize forage in strips along roads, around important vegetation stands, or on wildland/urban interface areas. For example, "firefighting sheep" are being used successfully to graze fuel breaks in cheatgrass-infested areas on the wildland interface in Carson City, Nevada (Anonymous 1999).

Herbicides

The use of herbicides to reduce fine fuels can be implemented without the soil surface disturbance associated with other methods such as grazing or mechanical fuel breaks. Economics, environmental impacts, selectivity, and effectiveness are among the factors that must be considered prior to selecting an herbicide to reduce fuels in cheatgrass-infested rangelands. Eckert et al. (1974) evaluated some of these criteria and identified Atrazine as an herbicide that can successfully control cheatgrass.

Recently, OUST® (sulfometuron methyl), a DuPont registered herbicide (DuPont 1996), has been used to reduce or eliminate cheatgrass prior to seeding perennial plants in fire rehabilitation or greenstripping projects (Pellant et al. 1999). OUST® has a short half-life, low toxicity, and is approved for use on public lands (USDI 1991). OUST® kills germinating plants by inhibiting amino acid biosynthesis necessary for meristematic growth (Kishore and Shah 1988). Therefore, OUST® can be used to control annual species such as cheatgrass, while causing little damage to established perennial plants. These perennial plants retain greenness and fire resistance longer into the fire season with the additional soil water and nutrients that are available after weed control.

Cost of treatment with OUST® (herbicide and application) is variable ($20 to $40 per acre), depending on the size of project and whether aerial or ground applications are used. One important restriction on the use of OUST® is that livestock must be excluded from the treated areas for 1 growing season following application. This restriction precludes the use of OUST® for fuel-break establishment in grazed pastures. Perhaps the best situation in which to use OUST® for fuel-break purposes in the NCA is along fenced road or highway rights-of-way where livestock are excluded.

Another potential strategy to reduce wildfires is the application of OUST® on cheatgrass-infested shrublands with remnant populations of native perennial plants. With the reduction of cheatgrass in the shrub understory, native perennial plants could increase in both numbers and vigor. Eventually the potential for wildfires would be reduced as perennial plants replace cheatgrass in the understory. However, the impacts of OUST® on the entire biota in native shrublands (e.g., soil microbes, microarthropods, insects, herbivorous rodents, passerine birds) are not well understood. Further studies are warranted prior to the operational use of this strategy.

Fire-Resistant Vegetation (Greenstrips)

Greenstrips are strips of fire-resistant vegetation placed at strategic locations on the landscape to slow or stop the spread of wildfires (Pellant 1990). The use of fire-resistant vegetation is not new (Platt and Jackman 1946) nor is it limited to the Intermountain area (Green 1977). Idaho BLM initiated the Greenstripping Program in 1985 (Pellant 1994), and several of the first greenstrips in Idaho were established in the NCA. The following benefits are expected with the successful establishment of a greenstrip network:

1. Reduction of wildfire encroachment into fire-susceptible shrublands.
2. Breaking up of large areas dominated by flammable annual grasses into more manageable blocks from a fire suppression perspective.
3. Reduction in fire suppression and rehabilitation costs.

Greenstrip width varies from 30 to 600 feet (9.1-182.4 m), depending on fire prevention objectives, topography, and soils. Weed control and site preparation (disking, burning, or herbicides) are essential prior to seeding (Hull and Stewart 1948, Hull and Holmgren 1964, Monsen 1994*a*). Plants used in greenstrips should have the following characteristics:

1. Fire resistant throughout the wildfire season and fire tolerant if burned in a wildfire.
2. Drought tolerant and adapted to persist on semi-arid sites in competition with weeds.
3. Palatable to herbivores, yet not susceptible to mortality with grazing.

A variety of plant materials meet these criteria and have been used for greenstripping in the NCA. Crested wheatgrass (*Agropyron cristatum*), Russian wildrye (*Elymus junceus*), and Siberian wheatgrass (*Agropyron sibiricum*) are the most commonly used grasses in greenstripping. These grasses, when well established, generally meet all of these criteria except fire resistance throughout the fire season. Alfalfa (*Medicago spp.*) is the only forb that has been used successfully in some NCA greenstrips. Shrubs are not generally used in greenstrips due to their flammability and fuel loading. However, forage kochia (*Kochia prostrata*), an introduced half shrub, meets all of the criteria and is a common component of many NCA greenstrips. Concerns about the invasiveness of forage kochia have limited its application in areas where sensitive plant species occur. Additional recommendations on plant materials used for greenstipping are found in Monsen (1994*b*) and Pellant (1994).

Establishment and persistence of seeded species in arid environments are uncertain (Jordan 1983) and influenced by the type of equipment used to prepare seedbeds and distribute seed, as well as by climate and site condition (e.g., burned versus unburned) (Monsen 1994*b*). Dry conditions from 1987 to 1989 caused seeding failures on many of the greenstrip projects established in the NCA. The NCA Management Plan (USDI 1995) recognized the utility of greenstrips and identified the reseeding of poorly established greenstrips as a requirement prior to establishing new greenstrips.

The success of greenstrips in reducing the spread of wildfires has not been well documented. Greenstrips inspected following wildfires in the NCA from 1994 to present were effective in reducing or stopping wildfire spread, especially in combination with additional fuel breaks (e.g., nearby roads) and limited fire suppression efforts. However, wildfires also have breached poorly established greenstrips that were dominated by cheatgrass. Additional evaluations are needed as wildfires contact greenstrips to better document the effectiveness of these living fuel breaks in reducing wildfire spread.

SUMMARY

There is not a single nor a simple fuels management solution to reducing wildfire impacts within the NCA. The environment and wildfire cycle have been permanently altered by the introduction and spread of cheatgrass throughout the 485,000-acre (196,425-ha) management area. Medusahead wildrye is becoming more common in parts of the NCA and presents even greater fire management problems, since slower decomposition rates compared to cheatgrass (due to high tissue silica content) result in fuel accumulation. Therefore, it is likely that the magnitude of the wildfire problem will increase in the future unless a proactive and effective fuels management program is designed and implemented in the NCA. Available fuels management options described above must be used singly or in combination to restore and maintain the natural values of the NCA.

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REHABILITATION EFFORTS, HISTORY, AND COSTS IN THE SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA

Steven Jirik

Since 1979, wildfire has destroyed approximately 2/3 of the shrub habitat of the Birds of Prey National Conservation Area (NCA). Consequently, much of the NCA now consists primarily of a cheatgrass-dominated landscape that provides poor wildlife habitat, increases fire frequency, and leaves remaining shrublands highly susceptible to future fires.

Drought and cheatgrass (*Bromus tectorum*) competition are the 2 biggest challenges to reestablishing perennial vegetation following wildfire. The climate of the NCA consists of a xeric moisture regime, mesic temperature regime, and only 7-10 inches (178-254 mm) average annual precipitation. During the 1986-1994 drought, some areas received less than 5 inches (127 mm) of precipitation annually. Seedings can be successfully established in years with adequate precipitation on sites where cheatgrass competition is minimal. This includes recently burned sagebrush (*Artemisia tridentata*) or winterfat (*Krascheninnikovia lanata*) sites that had well-developed biological soil crusts and minimal cheatgrass prior to burning. Cheatgrass-infested sagebrush stands that have burned intensely enough to kill most of the cheatgrass seed on the soil surface also can be seeded successfully. Seedings are seldom successful on previously burned, cheatgrass-dominated sites that no longer contain a sagebrush overstory. On these sites, soil surface temperatures during fire are insufficient to kill cheatgrass seed lying on the ground.

Prior to the 1991 "Decision for the Vegetation Treatment on BLM Lands in the Thirteen Western States," the use of herbicides to control cheatgrass was prohibited on public land. Therefore, various tilling methods such as plowing and disking were the only available options. Unfortunately, these treatments obliterated remaining native vegetation and biologic soil crusts, increased site susceptibility to wind erosion, often resulted in seed being drilled too deeply, and opened up the site for total cheatgrass domination if seedings were unsuccessful. Prescribed fire was used in attempts to kill cheatgrass seed still on the plant. Although some seeds were killed, the number of seeds remaining on the soil surface was adequate to fully occupy the site the following spring. Intensive livestock grazing also may reduce cheatgrass competition. Realistically, during the short period of time when cheatgrass is highly palatable, a sufficient number of livestock cannot be concentrated on a small enough area to reduce the cheatgrass significantly. In addition, this type of grazing can be detrimental to remaining perennial grasses.

In 1991, the vegetation treatment environmental impact statement (EIS) was approved for public lands in 13 western states, allowing BLM to use a limited number of herbicides for vegetation treatments. The herbicide OUST® (DuPont) is one of the chemicals approved for use and is effective in controlling annual grasses while having minimal impacts on most established perennial species. OUST® has a wide application window, from late fall through early spring. Residual action in the soil controls cheatgrass for 1 to 3 years, depending on soil moisture, pH, and temperature. It is classified as nontoxic to fish and wildlife.

Funds to treat cheatgrass-infested areas burned by wildfire must come from sources other than Emergency Fire Rehabilitation (EFR). In 1996, a special appropriation from Congress provided the funding to apply OUST® on 10,000 (4,050 ha) of the 50,000 acres (20,250 ha) that burned in the NCA that year, which was the largest fire year on record. Twenty thousand acres (8,100 ha) of recently burned sagebrush were seeded that fall. Ground-applied OUST® treatments began in late fall 1996 on areas where it was determined seeding would be unsuccessful without cheatgrass control. The acre limitations of ground spray equipment, equipment malfunction, wind, and contract default extended application into early May 1997. The OUST®-treated areas were drill seeded with perennial grasses in fall 1997 and aerially seeded with sagebrush and winterfat the following winter. In 1997, the BLM's Lower Snake River District (LSRD) obtained a tractor-mounted 3-point hitch sprayer with a Raven® Control System which can treat approximately 100 acres (40.5 ha) per day. This equipment has been highly effective and widely used for treating greenstrips and smaller fire rehabilitation and restoration projects. However, it is inadequate for treating large areas. In 1996, the LSRD made a request to the Idaho Department of Agriculture to

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allow aerial OUST® applications. DuPont developed a 24C Special Use Label which was approved by the State of Idaho for aerial applications. Aerial application allows treatment of up to 2,000 acres (810 ha) per day, greatly increasing efficiency on large projects. Product and application costs are about $30 per acre for both aerial and ground-spray treatments.

Before 1997, all habitat rehabilitation seedings were conducted following wildfire and funded with fire rehabilitation funds. Therefore, random fire occurrence determined where habitat rehabilitation efforts occurred. Recently, the LSRD has conducted some small-scale non-EFR restoration projects ranging in size from 150 to 1,000 acres (61 to 405 ha). Most commonly, rehabilitation efforts involve drill seeding perennial grasses in the fall, followed by aerially seeding shrubs later in the winter. Available plant materials that are adapted to the harsh environment of the NCA are limited. Perennial grasses used on most of the NCA include Siberian wheatgrass (*Agropyron sibericum*), Desert wheatgrass (*Agropyrum desertorum*), and Russian wildrye (*Psathyrostachys juncea*). The grass is seeded with a rangeland drill in the fall following fire and then aerially seeded with Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and winterfat in the winter on their respective sites. Forage kochia (*Kochia prostrata*) is often included in the seed mix on drier sites because it is highly palatable to wildlife, competes well with cheatgrass, retards the spread of fire, and resprouts if burned. Indian ricegrass (*Achnatherum hymenoides*) and sand dropseed (*Sporobolus cryptandrus*) are often used on sandy sites.

Before 1993, Fairway and Hycrest crested wheatgrass (*Agropyron cristatum*) were seeded extensively on EFR seedings. However, nearly all these grasses died during the 1986-1993 drought. Before 1983, policy constraints precluded the use of shrubs in EFR seedings. In 1983, the Idaho State Director authorized the use of fourwing saltbush (*Atriplex canescens*) and winterfat in EFR seedings. Because winterfat's fluffy seeds caused broadcast seeders to clog, district personnel devised a number of ways to alleviate this problem, including hydroseeding and coating the seed with clay so it would pass through broadcast seeders. Later, seed companies refused to coat the seed because of the mess involved. Hydroseeding was extremely time consuming and labor intensive. Eventually, various contractors developed the technology to broadcast unaltered winterfat seed. Wyoming big sagebrush was first seeded in the 1986 Initial Point EFR. At that time, only enough seed was available to treat 10 acres (4 ha).

Currently, native plant materials are used when they are available and adapted to local ecological sites. Wyoming big sagebrush is seeded on loamy 7-10" ecological sites, and winterfat is seeded on silty 7-10" ecological sites. Because native NCA winterfat seldom produces sufficient quantities of viable seed, a central Utah selection is used. Secar Snake River wheatgrass (*Elymus wawawaiensis*) and thickspike wheatgrass (*Elymus lanceolatus*) are seeded on the few loamy 10-12" (254-305 mm) ecological sites, e.g., near Kuna Butte in the northwestern corner of the NCA. Indian ricegrass and sand dropseed are often seeded on sandier sites such as the sandy loam 7-10" ecological site. Although available, Indian ricegrass is expensive and usually limited to seedings of less than 300 acres (122 ha). Table 1 summarizes the cost of native seed mix compared to a "typical" mix used for post-fire rehabilitation.

Table 1. Seed cost comparison of an all-native mix for sandy soils with that of a native/nonnative mix (June 1999).

Native Sandy Mix				**Native / Non-Native Mix**			
Species	**Lbs/acre[a] (bulk)**	**Cost/lb**	**Cost / acre**	**Species**	**Lbs/acre[a] (bulk)**	**Cost/lb**	**Cost/acre**
Indian ricegrass	6.0	$11.20	$67.20	Siberian wheatgrass	4.0	$ 2.00	$ 8.00
Sand dropseed	1.0	$ 4.10	$ 4.10	Russian wildrye	2.0	$ 4.00	$ 8.00
Winterfat	1 (0.3 pls)	$10.00	$10.00	Winterfat	1 (0.3 pls)	$10.00	$10.00
Wyoming sagebrush	1 (0.1 pls)	$ 4.00	$ 4.00	Wyoming sagebrush	1 (0.1 pls)	$ 4.00	$ 4.00
Fourwing saltbush	1.0	$ 5.00	$ 5.00				
Total	**10.0**		**$90.30**	**Total**	**8.0**		**$ 30.00**

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[a] Metric conversions: 1 lb/ac = 1,122 g/ha

Most of the NCA consists of the loamy 7-10" (Wyoming big sagebrush / Thurber needlegrass) ecological site. Under presettlement, Thurber needlegrass (*Achnatherum thurberianum*) was the dominant grass in the NCA. Unregulated livestock grazing during the late 19th and early 20th centuries extirpated this plant from most of the lower Snake River Plain. Thurber needlegrass seed is only available in very small quantities obtained from wildland collections and is extremely expensive. Bottlebrush squirreltail (*Elymus elymoides*) and Sandberg bluegrass (*Poa secunda*) are the 2 remaining native perennial grasses still common in the NCA. Available plant materials for these species have exhibited poor success on NCA test plots. They are also expensive and limited in their availability. Table 2 compares the cost of using all native plant materials (assuming they are available) versus a typical native/nonnative mix on a loamy 7-10" ecological site.

Table 2. Per-acre seed cost comparison (June 1999) of a typical NCA seeding and an all-native seeding on a loamy 7-10" (Wyoming big sagebrush / Thurber needlegrass) ecological site.

Typical Native / Nonnative Mix				**All-Native Mix**			
Species	**PLS lb/acre**	**PLS cost/lb**	**Cost/ acre**	**Species**	**PLS lb/acre**	**PLS cost/lb**	**Cost/ acre**
P27 Siberian Wheatgrass	6.0	$ 1.20	$ 7.20	Thurber Needlegrass	2.0	$50.00	$100.00
Bozoisky Russian Wildrye	2.0	$ 3.50	$ 7.00	Bottlebrush Squirreltail	2.0	$18.00	$ 36.00
Wyoming Big Sagebrush	0.1	$40.00	$ 4.00	Sandberg Bluegrass	4.0	$ 5.50	$ 22.00
				Wyoming Big Sagebrush	0.1	$40.00	$ 4.00
Total			**$18.20**	**Total**			**$162.00**

The high cost, lack of availability, and poor success of native grass seed currently limits its use in the NCA. To increase the availability of adapted native seed, the BLM and Forest Service Shrub Sciences Lab (FSSSL) are collaborating with various seed growers by providing Thurber needlegrass, bottlebrush squirreltail, and showy penstemon (*Penstemon speciosus*) seed that was originally collected from within or near the NCA. The growers will develop technology to increase and harvest seed. BLM will create a demand for the seed by guaranteeing a specific price for seed produced in the next few years. In return, the growers will share their technology with other growers. Providing seed to growers directly from wildland collections will help decrease the time needed to get source-identified native seed on the market as opposed to the more time-consuming traditional seed certification process.

In addition to working with native seed growers, BLM has established 3 demonstration plots to test the long-term success of various native plant materials for potential future use in the NCA. These materials were donated by various Natural Resources Conservation Service Plant Materials Centers, and the FSSSL. Additional test plots will be planted in 1999 with forb seedlings such as Munro's globemallow (*Sphaeralcea munroana*), gooseberry leaf globemallow (*S. grossulariifolia*), showy penstemon, arrowleaf balsamroot (*Balsamorhiza sagittata*) and oval-leaf buckwheat (*Eriogonum ovalifolium*).

Poster Presentations: Abstracts

SAGEBRUSH STEPPE ECOSYSTEMS SYMPOSIUM
Boise State University
June 21-23, 1999

POSTER PRESENTATIONS

Interactive Multimedia Computer Presentations for Land Management Agencies

Gary O. Grimm and Katy Flanagan
Mountain Visions, Boise, Idaho

With the Bureau of Land Management and other federal, state, and local agencies, Gary O. Grimm and Katy Flanagan of Mountain Visions, a Northwest multimedia consortium based in Boise, Idaho, have been developing unique computer multimedia presentations. They demonstrated a recent production for the U.S. Department of the Interior – The Aurora Project, Community Watershed Partnerships. This is an interactive multimedia documentary produced for CD-ROM, the Internet, and Computer Kiosk use. This remarkable immersive virtual adventure prototypes a new approach to understanding the dynamics of landscape and watershed restoration. Choose a spot high above the Western U.S. and “fly” into a location via an actual 3-dimensional map animation. Click on a site map and you are immersed in one of several panoramic watershed areas. Each location includes 360-degree landscape panoramas, numerous embedded audiovisual and video hot spots, interviews, narratives, and natural sounds. Enjoy a virtual exploration of the landscape, via your personal computer, while you discover flora and fauna and the dynamics of rivers, streams, riparian habitats, and complex watershed drainages. The 360-degree interactive panoramas and/or moving digital video can be used for monitoring natural or man-made changes in the environment. For orientation purposes, the use of a rotating 360-degree compass arrow and overview maps accompany the panoramas. Also demonstrated are techniques to make simplified computer multimedia layered-map presentations from complex Geographical Information System (GIS) data.

Interior Columbia Basin Ecosystem Management Project: Regional Implementation Support Team

Richy Harrod, Fay Shon, and Al Horton
Interior Columbia Basin Ecosystem Management Project

This poster announces the capabilities and availabilities of 2 interagency, interdisciplinary technology transfer teams for the Interior Columbia Basin Ecosystem Management Project (ICBEMP). These teams, known as the Regional Implementation Support Teams (RISTs), are charged with facilitating and supporting the transfer of information contained in the science documents of the ICBEMP to field organizations who are actively working on field projects. RISTs are drawn from U.S. Forest Service and Bureau of Land Management subject-matter specialists. Subject areas include:

- Forest Ecology
- Range Ecology
- Aquatic Ecology
- Hydrology
- Terrestrial Biology
- Fire Ecology
- Socio-economics
- Soils
- Data Support
- Modeling
- Recreation
- Planning
- Cultural/Tribal Issues

Teams can be customized to meet both short-term training or long-term consultation needs of field project managers. Public land managers may inquire further or arrange for consultations through either of the 2 team leaders listed below:

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Restoration of the Moss Component of Microbiotic Crust to the Western Snake River Plain, Idaho

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Microbiotic soil crusts serve several roles important to proper function of the arid and semiarid ecosystems where they occur. Conversion of native sagebrush steppe vegetation to grasslands composed of mostly invasive exotic annual plants results in degradation or loss of microbiotic crusts. Investigating possibilities for restoration of microbiotic crusts is an essential step in reclaiming functional complexity of the native ecosystem. Fragmented gametophytic thalli of 3 arid-land mosses, *Bryum argenteum* Hedw., *Ceratodon purpureus* (Hedw.) Brid., and *Tortula ruralis* (Hedw.) Gaertn., Meyer & Scherb., were used in laboratory and field experiments to determine potential for restoring perennial moss growth to sites devoid of them and composed of exotic annual grassland. Laboratory experiments yielded positive results for use of these mosses in restoration efforts. Field experimentation provided insight about treatments appropriate for the preparation of exotic annual grassland for inoculation with moss fragments.

Know Your Squirreltail Taxa

T.A. Jones and D.C. Nielson
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In the definitive taxonomic treatment, Wilson (1963) recognized 4 squirreltail (*Elymus* spp. = *Sitanion* spp.) species, including 1 with 2 infraspecific taxa. All taxa (*hystrix*, *californicum*, *jubatum*, *longifolium*, and *hordeoides*) are found in southern Idaho, but the ecological amplitude of each is poorly understood. These taxa are easily identified by examining the inflorescence, particularly the awns. Again, the genetic relationships between the taxa are poorly described, but all reports of chromosome number are 2n=4x=28. Based on floral morphology, Wilson believed that *longifolium* was the most primitive taxon of the group. *Jubatum* is more common in southwestern Idaho than to the east. *Longifolium* is most common above the Snake River Plain. *Hystrix* is the predominant taxon in the most arid areas of southern Idaho. *Californicum* and *hordeoides* are the least common in southern Idaho, while *longifolium* and *jubatum* are the largest statured. *Longifolium* inflorescences disarticulate primarily at the base of the spike, while *hystrix*, *californicum*, and *jubatum* disarticulate at every node in the spike. Among southern Idaho collections, *longifolium* is later than *hystrix* but earlier than *jubatum* when grown at a common site. Multiple taxa may be found at the same site, and they can often be easily spotted in the field by their contrasting maturity and stature.

Monitoring the Rehabilitation Treatments for the Eighth Street Fire: A Coordinated Effort

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Dr. Frederick Pierson, Hydrologist, NW Watershed Research Center, USDA Agricultural Research Service, Boise, Idaho
Lynn Wessman, Ecologist, Lower Snake River District, Bureau of Land Management, Boise, Idaho

In August 1996, the Eighth Street Fire burned more than 15,000 acres (6,075 ha) in the foothills adjacent to Boise. In the wake of this fire, federal, state, county, and city agencies all participated in a massive rehabilitation effort that included soil stabilization and reestablishment of vegetation. An interdisciplinary team developed a monitoring plan which identified 3 separate work groups. One group dealt with the response of vegetation, 1 with the effectiveness of soil stabilization treatments, and 1 with fire and treatment effects on infiltration, runoff, and erosion. The individuals who participated on the work groups were professionals from Boise National Forest, Natural Resources Conservation Service, Idaho Department of Lands, Idaho Department of Fish and Game, Bureau of Land Management, Agricultural Research Service, and USDA Forest Service Rocky Mountain Research Station. The public's continued interest and involvement in the Eighth Street fire has created a unique opportunity to implement a long-term watershed monitoring program. The product of the 5-year assessment will generate useful information for planning and implementing future fire rehabilitation efforts. This poster presents the structure, objectives, and methods used and some preliminary results 2 years following the fire.

Biological Soil Crusts: Natural Barriers to *Bromus tectorum* L. Establishment in the Northern Great Basin, USA

Julienne H. Kaltenecker, Marcia C. Wicklow-Howard, and Kelly Larsen
Boise State University, Boise, Idaho

Mike Pellant, Bureau of Land Management, Idaho State Office, Boise, Idaho

In 1993, studies were initiated to investigate the relationship between *Bromus tectorum* L., an invasive alien annual grass, and biological soil crusts (BSCs), a conspicuous component of *Artemisia* shrub steppe communities in the northern Great Basin. Observations indicated that sites with undisturbed BSC cover had low *B. tectorum* despite nearby seed sources. BSCs were fragmented in random plots and left intact (as a control) in others. The following year, *B. tectorum* densities in the fragmented plots were almost twice that of the control, indicating some barrier to establishment. A subsequent study addressed BSC recovery following wildfire, a factor that contributes to *B. tectorum* invasion. Cover of *B. tectorum*, perennial vegetation, and BSCs were measured in sites that burned in past summer wildfires. Each site contained 2 post-fire treatments: 1) seeded with perennial grasses, and 2) control (no treatment). The controls were dominated by *B. tectorum.* BSCs in the control consisted of 5 moss species and 7 lichens and occurred only in pockets of low *B. tectorum* cover. BSCs in the seeded treatments were more diverse, containing 8 mosses and 17 lichens, and formed a dense, continuous carpet between perennial plants. *Bromus tectorum* cover was negligible. Recovery (natural or artificial) of the native community structure with open spaces between patchy perennial plants appears to enhance recovery of BSCs in terms of both cover and diversity. BSCs may then provide long-term protection against *B. tectorum* encroachment.

Case Studies in Arid Land Restoration

Ed Kleiner, CEO, Comstock Seed, Reno, Nevada

This poster board is a summary of 8 reclamation/restoration projects in the western United States, including mines, highways, and utility corridors. Except for the Leviathan project, they are all private ventures. My data regarding project installation is fairly complete. However, my access to quantified data regarding project performance varies greatly. The results presented here are primarily visual. These projects typically are bonded, and quantified results will be necessary to get bonding release. Also, these projects are cross-referenced for both similar and contrasting results that I found informative for adding depth to our reclamation perspective. I am hesitant to draw generalized conclusions or to create generic formulas for reclamation programs. From the examples, one can see that the approaches to soil conditioning and seeding techniques are as varied as the habitat conditions. While some clients have relied exclusively on organic amendments, others have emphasized inorganic fertilizers. The common threads to all sites are concern for the growing medium, the presence of some program to improve the soil condition prior to seeding, and an emphasis on native species. I think the primary causes for project failure include lack of a soils program, incorrect seed application, and the vagarious nature of climate.

Assessing and Monitoring Habitat Integrity of *Lepidium papilliferum* (Slick-spot Peppergrass) in the Sagebrush Steppe of Southwestern Idaho

Michael Mancuso, Robert Moseley, and Christopher Murphy
Idaho Department of Fish and Game, Conservation Data Center, Boise, Idaho

Slick-spot peppergrass (*Lepidium papilliferum*) is a rare plant endemic to the western Snake River Plain and adjacent foothills in southwestern Idaho. It is restricted to visually distinct, small-scale openings created by unusual edaphic conditions within the regional sagebrush-steppe ecosystem. Widespread habitat degradation, fragmentation, and conversion have occurred throughout the species' range. Many populations have been extirpated during the last century, and the long-term prospects for many extant populations are grim. As a result, slick-spot peppergrass is 1 of

Idaho's highest-priority plant conservation concerns. We have developed a Habitat Integrity Index to assess and monitor the ecological integrity of slick-spot peppergrass habitat. Habitat monitoring focuses on the most important factor responsible for the decline of slick-spot peppergrass, namely, the loss of high-quality shrub-steppe vegetation. Metrics for the Index use physical features, community composition, and community structure attributes to rate occurrence integrity on a relative scale. Attributes focus on wildfire, livestock grazing, and off-road motorized disturbances. All 3 are widespread, interrelated, and management concerns in the sagebrush steppe of southwestern Idaho. In 1998, baseline Habitat Integrity Index data were collected at 37 extant slick-spot peppergrass occurrences. Information from the Index will be used to monitor long-term trends regarding habitat quality and species conservation. The conservation of slick-spot peppergrass is largely dependent on conserving its sagebrush steppe habitat. Ideally, the Index can be 1 part of a more comprehensive conservation approach on behalf of southwestern Idaho's shrub steppe ecosystem.

Seasonal Nutrient Dynamics of Five Forage Shrubs and One Perennial Grass in a Cold Desert Ecosystem

Kelly L. Memmott and Stephen B. Monsen, USDA Forest Service
Rocky Mountain Research Station, Intermountain Shrub Science Lab, Provo, Utah

Val Jo Anderson, Brigham Young University, Provo, Utah

Variability in forage nutrient levels differ by species and time of season. In this study, the nutrient levels of the shrubs prostrate kochia, fourwing saltbush, rubber rabbitbrush, big sagebrush, and winterfat were determined throughout a growing season. The study site was adjacent to the Brigham Young University Skaggs Research Ranch, near Malta, Idaho. These 5 shrubs were planted as seedlings in 1.5-m tilled strips into established crested wheatgrass pastures. Eight years after transplanting, nutrient status of the grass/shrub matrices was monitored at intervals of 2 or 3 weeks from the beginning of shoot development in May 1993 until snowfall in December 1993. All samples were evaluated in a full suite nutrient analysis. Significant differences were found between nutrient status of crested wheatgrass and the 5 shrubs as the season progressed. Percent crude protein was significantly higher for all shrubs (range of 27.15 to 9.11%) than for crested wheatgrass (range of 13.15 to 3.79%) throughout the growing season. This trend held true for total digestible nutrients, phosphorus, calcium, digestible dry matter, and metabolizable energy.

Development of Site-adapted Ecotypes of Bluebunch Wheatgrass, Sandberg Bluegrass, and Thurber Needlegrass for Restoration of Sagebrush Steppe Communities on the Snake River Plain

Stephen B. Monsen, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah

Our ability to restore disturbed sagebrush steppe communities is currently limited due to inadequate seed supplies of site-adapted native species. Important cultivars of a number of species native to the Intermountain region have been developed and are currently sold by commercial seed companies. However, few species native to the Snake River Plain are commercially harvested or produced in sufficient quantities to support large restoration projects. Studies were initiated in 1988 and 1989 to assemble and evaluate collections of bluebunch wheatgrass (*Pseudoroegneria spicata*), Sandberg bluegrass (*Poa secunda*), and Thurber needlegrass (*Achnatherum thurberianum*) from the Plain and surrounding locations. These collections were evaluated to identify individual ecotypes and their distributions. Persistence, growth habit, seed production, and competitive attributes of field plantings established near Boise, Idaho, were evaluated over a 10-year period. One selection of bluebunch wheatgrass from Anatone, Washington, has excelled in seedling establishment, competitiveness, and overall adaptability to conditions in the Snake River Plain. This selection yields an abundance of large seeds that produce vigorous seedlings. It is currently being grown under field conditions to support a Source-Tested germplasm release in 1999. Little variability existed among populations of either Sandberg bluegrass or Thurber needlegrass from the Snake River Plain. Although differences in plant stature and seasonal periods of growth occurred among collections of both species, differences are not sufficient to recommend more than 1 germplasm for plantings in disturbed sites on the Plain. A collection of Sandberg bluegrass acquired near Mountain Home, Idaho, will be released in 1999 as a Source Identified germplasm. A Source Identified germplasm of Thurber needlegrass acquired near Orchard, Idaho, will also be released in 1999. Seeds of all 3 releases will be available to commercial growers through the Utah Crop Improvement Association.

Use of OUST® Herbicide to Control Cheatgrass in the Northern Great Basin

Mike Pellant and Steven Jirik, Bureau of Land Management, Boise, Idaho
Julienne H. Kaltenecker, Boise State University, Boise, Idaho

The invasion and dominance of native rangelands with cheatgrass (*Bromus tectorum*), an exotic annual grass, has significantly disrupted ecological processes and increased extent and frequency of wildfires in the northern Great Basin. Traditional cheatgrass control measures include prescribed fire, mechanical control, grazing, and herbicides. The use of herbicides for cheatgrass control was prohibited until July 1991, when an Environmental Impact Statement on "Vegetation Treatment on BLM Lands in the Western States" was approved. OUST®, a registered DuPont herbicide (sulfometuron methyl), inhibits apical meristem growth and is particularly effective on weedy annuals with little adverse impact to established perennial species. It is applied as a liquid spray and functions as both a pre- and post-emergent herbicide. OUST® was compared with burning and disking to control cheatgrass on research plots in northern Nevada in 1992. Results indicated that OUST® provided the most effective control of cheatgrass (26% frequency of occurrence) compared to the disking, burning, and control treatments, 46%, 42%, and 72% frequency of occurrence of cheatgrass, respectively. In April 1995, OUST® was applied operationally to a 100-acre cheatgrass-infested seeding near Mountain Home, Idaho. Compared to an adjacent untreated control, the OUST® treatment reduced cheatgrass density by 91%. Remnant perennial grasses were much more vigorous (biomass and seedstalk production) in the OUST®-treated area compared to the control. Results from these studies indicate that OUST® can be used to effectively control cheatgrass at a cost of around $20 - $25/acre (herbicide and application costs).

Cheatgrass Expansion and Biodiversity Loss on the Snake River Plain, Southern Idaho

Victoria Saab and Nancy L. Shaw, USDA Forest Service
Rocky Mountain Research Station, Boise, Idaho

Stephen B. Monsen, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah

Terry Rich, USDI Bureau of Land Management, Washington Office, Boise, Idaho

Cheatgrass (*Bromus tectorum*) was introduced into the western United States early in the 20th century and spread rapidly to occupy degraded shrub steppe communities in the Snake River Plain. Cheatgrass competition and subsequent increases in fire frequency have contributed to further decline of native flora and fauna. Antelope bitterbrush (*Purshia tridentata*), basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*), Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*), and salt desert shrub communities have been replaced by this annual grass. Alterations of these plant communities have adversely affected obligate shrub steppe animal species; e.g., sage grouse (*Centrocercus urophasianus*), Paiute ground squirrel (*Spermophilus mollis*) (formerly Townsend's ground squirrel [*Spermophilus townsendii]*), and Brewer's sparrow (*Spizella breweri*) have experienced population declines. Long-term succession studies demonstrate that recovery of cheatgrass-dominated communities proceeds slowly or possibly not at all when native seed sources are lacking. Expansion of cheatgrass continues into more xeric communities through development of adaptive ecotypes and by expansion into areas affected by fire disturbances. In addition, more troublesome perennial weeds are now displacing cheatgrass over portions of its range. As a result, individual species are imperiled or lost, plant and animal diversity is reduced, and the ecosystem becomes simplified and less resilient. Without large-scale active restoration, these ecosystems, with their associated flora and fauna, are at risk.

Techniques for Reestablishment of Cool-Season Grasses

Nancy L. Shaw, USDA Forest Service, Rocky Mountain Research Station, Boise, Idaho
Stephen B. Monsen, USDA Forest Service, Rocky Mountain Research Station, Provo, Utah

Improving biodiversity on annual rangelands requires reestablishment of native perennial grasses. We conducted studies to examine the effects of (1) 3 seeding rates: 66, 330 and 1,320 seeds/m, and (2) 3 seeding methods: broadcasting, single-row seeding, and double-row seeding (1,650, 1,650, and 3,300 seeds/m^2) on establishment of Snake River wheatgrass (*Elymus wawawaiensis*), Sandberg bluegrass (*Poa secunda*) and bottlebrush squirreltail (*Elymus elymoides*) on cheatgrass (*Bromus tectorum*)-dominated sites. Plots for each study were established and seeded in fall 1994 at Orchard and at the Shrub Garden in southwestern Idaho. After 2 years, density of all grasses when seeded at the low rate was 5 plants/m at Orchard; densities were 27/m for Snake River wheatgrass and 2/m for Sandberg bluegrass and bottlebrush squirreltail at the Shrub Garden. Density of Snake River wheatgrass and Sandberg bluegrass was improved 4 and 7 times by seeding at the moderate rate at the Shrub Garden and 3 to 26 times by seeding at the high rate at both sites. Bottlebrush squirreltail did not respond to seeding rate. After 2 growing seasons, seeding method had no effect on seeded grass density at Orchard or on bottlebrush squirreltail density at the Shrub Garden. Density of Snake River wheatgrass plants at the Shrub Garden was 460/m^2 when single- or double-row seeded, compared to 152/m^2 when broadcast. Establishment of Sandberg bluegrass was 86/m^2 when single-row seeded or broadcast, but increased to 244/m^2 with double-row seeding. Low survival on some treatments and poor development of most seeded grasses indicates seeding technology that physically separates seeded species from annual competition be used in conjunction with appropriate seeding methods and rates to provide favorable seedbed conditions for seedling establishment.

PLATEAU®: A New Product for Leafy Spurge (*Euphorbia esula*) Control

Joseph G. Vollmer and Jennifer L. Vollmer, American Cyanamid Company, Boise, Idaho

Leafy spurge (*Euphorbia esula*) competes by shading, usurping available water and nutrients, and exuding plant toxins that prevent growth of other plants in the vicinity. Plant diversity is lost in these infested communities, along with a loss of wildlife habitat and reduction in grazing and land value. This deep-rooted perennial is difficult to control because of adventitious buds that are released when the top growth of the plant is injured. Buds have been known to be released several feet under the soil surface to emerge and replace the controlled growth. Typical herbicides used for leafy spurge control cause injury to or kill desired forbs. Areas are often treated for several years to achieve complete control. These treated areas eventually become grass monocultures due to the standard herbicide's inability to be selective between broadleaf species. PLATEAU® (imazapic) herbicide has the ability to translocate deeper in the root than the standard herbicides and is selective between broadleaf species. A trial was conducted in Theodore Roosevelt National Park to evaluate the efficacy of an aerial application of PLATEAU® herbicide for control of leafy spurge and to determine cottonwood tolerance to adjacent and direct spray applications. Tolerance of forb species was also noted to evaluate the selective control of PLATEAU® herbicide. Fall aerial PLATEAU® treatments resulted in 98% control of leafy spurge at both the 8 oz and 12 oz/acre (560 g and 841 g/ha)rate. Broadleaf species that survived the broadcast application without injury were cottonwood, snowberry, big sage, fringed sage, boxelder, green ash, western juniper, lupine, prairie scurf pea, and vetch.

Precipitation-Plant Community Production Covariation in Relation to Species Richness Within Sagebrush Steppe of Southern Idaho

Terence P. Yorks, Yong-Hong Li, and Neil E. West
Department of Rangeland Resources, Utah State University, Logan, Utah

Species richness was related to the fluctuation (F) of plant community production (P) in response to annual variations in precipitation (R), both separately and combined into an FPRI (covariance index [I]), using 10 years of end-of-growing-season phytomass data from 20 plots on each of 13 ungrazed relict sagebrush steppe sites in southern Idaho. A statistically significant ($p < 0.01$) relationship was observed between: overall mean above-ground production and crop-year precipitation, mean above-ground production and species richness, and mean above-ground production and FPRI. However, the slope of the precipitation-related aggregate regression was affected by shrub production being not straightforwardly correlated to crop-year precipitation inputs. Species richness, therefore, emerged as the clearest predictor among communities for consequent overall production. As an element of this, species-rich communities had more compensatory responses among individual species production following rainfall fluctuations than communities with lower species richness. In particular, the degree of compensation within species production between a favorable year (1963) and a drought year (1966) was significantly ($p = 0.02$) and positively related to species richness in the drought year. Nevertheless, compensation can only explain part of the pattern of plant community FPRI. Species appear intrinsically different in their constancy of response to the same precipitation. The proportion of perennial grasses and all annuals increased with overall production, so that FPRI generally increased with both community production and species richness. These results overall support, but do not confirm, hypotheses suggesting that species diversity begets community productivity and stability.

Sagebrush Steppe Ecosystems Symposium Field Tour Summary

SAGEBRUSH STEPPE ECOSYSTEMS SYMPOSIUM
Field Tour

Snake River Birds of Prey National Conservation Area
June 23, 1999

Stop 1: Wyoming big sagebrush relict area and adjacent burned cheatgrass-dominated site

The purpose of this stop was to view a relict sagebrush site, as well as a site where the sagebrush community had been replaced by cheatgrass as a result of wildfire. Roger Rosentreter, BLM Botanist, and Julie Kaltenecker, BSU Research Associate, discussed the plant and biological soil crust communities; Steve Knick, USGS Research Wildlife Biologist, discussed the various birds that inhabited the 2 sites.

Relict Site: This relatively undisturbed sagebrush/perennial grass stand had not received significant livestock grazing in over 20 years. Thurber needlegrass was removed by unregulated grazing during the early part of the century. Bottlebrush squirreltail and Sandberg bluegrass are the remaining perennial grasses. The area contains a well-developed biological soil crust, which is indicative of the low disturbance levels. Presence of the soil crust may contribute to the lack of cheatgrass in this area. Passerine birds breeding on this site, including sage thrashers, Brewer's sparrows, and sage sparrows, depend on shrublands.

Cheatgrass site: This site burned in the past and consists primarily of cheatgrass. The site still contains Sandberg bluegrass, which is hidden by the dominant cheatgrass. The site contains little biological soil crust due to high plant densities and litter accumulation and is highly susceptible to future wildfires. Passerine birds at this site include horned larks and western meadowlarks, which are associated with grasslands or disturbed landscapes.

Stop 2: 1995 Point Fire Rehabilitation (EFR) Seeding and Swan Falls Greenstrip

Point EFR Seeding: This site burned as part of the July 1995 Point wildfire. Bill Casey, BLM Fire Management Officer, discussed the 11,000-acre (4,455-ha) wildfire, which resulted in the deaths of 2 volunteer firefighters. A combination of high winds, extreme burning conditions, inadequate experience and training, and equipment failure all contributed to these fatalities. Steve Jirik, BLM Range Management Specialist, described the vegetation community that existed prior to the wildfire – a dense Wyoming big sagebrush stand with a dominant cheatgrass understory. He explained that the sagebrush burned hot enough to kill most of the cheatgrass seed lying under the sagebrush canopy and, as a result, herbicide was not needed to control subsequent cheatgrass growth.

The following species were seeded on the site:

Species	Variety	Lbs/acre (bulk)[a]	Method	Date
Siberian wheatgrass	P-27	3.5	rangeland drill	9/95
Standard wheatgrass	Nordan	3.5	rangeland drill	9/95
Wyoming big sagebrush	local	1.0 (0.1 pls)	aerial	1/96
Alfalfa	Ladak	1.0	aerial	1/96
		8.0		

Abundant moisture fell in 1996, with much of it occurring in early summer. Because the cheatgrass had already cured by that time, the seeded species had more available moisture, which allowed good establishment. However, cheatgrass still occupied most of the site in 1996 and 1997. In 1998, the seeded perennials began to out-compete the cheatgrass.

Swan Falls Greenstrip: Mike Pellant, BLM Ecologist, discussed BLM's greenstrip program in general and the Swan Falls Greenstrip in particular. The Swan Falls Greenstrip, which was plowed and drill seeded in fall 1989, failed because of drought and became invaded with cheatgrass. In spring 1994, the area was burned by prescription to remove cheatgrass seed and was drill seeded that fall with Siberian wheatgrass and Russian wildrye. The prescribed burn did not control the cheatgrass, which dominated the site the following spring. In fall 1996, the greenstrip was treated with OUST® herbicide at 1.0 oz/ac[b]. The herbicide application eliminated the cheatgrass and dramatically released the remnant seeding. However, the greenstrip was overseeded in fall 1997 to fill the vacant areas in the seeding. The disturbance from the drills allowed cheatgrass and tumblemustard to reinvade the site. The greenstrip was treated again with OUST® at 1.0 oz/ac in fall 1998 to release the existing seeding.

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[a] Metric conversion: 1 lb/ac = 1,122 g/ha

Stop 3: Dedication Point

John Sullivan, NCA Manager, discussed the overall purpose for and management of BLM's Dedication Point visitor education and interpretive site. The site, which was established 20 years ago, contains native winterfat and big sagebrush communities as well as areas where native species have been eradicated by wildfire and replaced with cheatgrass.

1999 Dedication Point Restoration Seeding: Steve Jirik discussed the restoration of a 140-acre site that burned in the early 1980s. The site, which consisted primarily of cheatgrass with some remnant perennial grass species, was burned by prescription in September 1998 to remove cheatgrass litter and was sprayed the following month with OUST® herbicide at 1.0 oz/ac to control subsequent cheatgrass growth. The following native species were seeded with a Truax-type drill in fall 1999:

Species	Lbs/acre (bulk)[2]	Cost/lb	Cost/acre
Indian ricegrass	6	11.20	67.20
Sand dropseed	1	4.10	4.10
Winterfat	1.0 (0.3 pls)	10.00	10.00
Wyoming big sagebrush	1.0 (0.1 pls)	4.00	4.00
Fourwing saltbush	1	5.00	5.00
Showy penstemon*	0.5	*	*
	10.5		$90.30
			$25.00 (labor/equipment)
			$115.30

*The penstemon seed was harvested from the NCA by volunteers.

An adjacent 2-acre demonstration plot was seeded with a durable seed mix to establish a comparison to the above native seeding. The species will include:

Species	Lbs/acre(bulk)	Cost/lb	Cost/acre
Winterfat	0.5 (.15 pls)	10.00	5.00
Wyoming big sagebrush	1.0 (0.1 pls)	4.00	4.00
Vavilov Siberian wheatgrass	4	2.00	8.00
Bozoiski Russian wildrye	2	4.00	8.00
Forage kochia	1.0 (0.1 pls)	4.00	4.00
	8.5		$29.00
			$25.00 (labor/equipment)
			$54.00

Dedication Point Overlook: Mike Kochert and Karen Steenhof, USGS Research Wildlife Biologists, discussed NCA research findings, including raptor responses to habitat alteration. Studies of raptors have been conducted in this area since the early 1970s. The density of nesting raptors in the NCA is higher than that recorded anywhere else in the world.

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[b] Metric conversion: 1 oz/ac = 70 g/ha

Snake River Birds of Prey NCA
Habitat Restoration Workshop
Field Tour Summary

SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA (NCA) HABITAT RESTORATION WORKSHOP

Field Tour

June 23, 1999

Note: This portion of the field trip occurred in the afternoon, immediately following the Sagebrush Steppe Ecosystems Symposium field trip and was attended only by those scientists and land managers participating in the NCA Habitat Restoration Workshop, which commenced the following day.

Stop 1: 1994 Poen EFR and Native Plant Materials Test Plot

Steve Jirik, BLM range management specialist, discussed the rehabilitation of the 550-acre Poen wildfire. The area, which is a loamy 7-10" Wyoming big sagebrush / Thurber needlegrass (ARTRW/STTH2) ecological site, supported a relatively good preburn stand of Wyoming big sagebrush. During the rehabilitation, the following species were seeded:

Species	Variety	Rate (lb/ac)[a]	Method	Date
Bluebunch wheatgrass	Secar	2.1	rangeland drill	1/94
Siberian wheatgrass	P-27	1.1	"	"
Standard wheatgrass	Nordan	1.1	"	"
Russian wildrye	Bozoisky	3.1	"	"
Fourwing saltbush	local	0.8	"	"
Forage kochia	Immigrant	0.3 (pls)	"	"
Winterfat	Hatch	1.5	"	"
Alfalfa	Ladak	1.5	legume box on drill	11/94
Wyoming big sagebrush	local	0.3 (pls)	aerial	1/95
Forage kochia	Immigrant	0.5 (pls)	"	"
Winterfat	Hatch	0.6 (pls)	"	"
Alfalfa	Ladak	1.2	"	"
		14.1 lb/ac		

The seeding was highly successful, with grass seedheads developing the first year (1995). However, the seeding was subsequently burned in 1996 by the Swan Falls wildfire, which killed most of the sagebrush, winterfat, and fourwing saltbush seedlings. The forage kochia and perennial grasses recovered the following year and have continued to grow into a mature stand.

John Doremus, BLM Wildlife Biologist, discussed his observations on the use of nonnative seeded species by Paiute ground squirrels (*Spermophilus mollis*) (formerly Townsend's ground squirrels [*Spermophilus townsendii*]), black-tailed jackrabbits, pronghorn, mule deer, and various songbirds. In nonnative seedings that lack native shrubs, such as this one, breeding songbirds are limited to only a few species (i.e., horned larks, western meadowlarks, and long-billed curlews).

Native Plant Materials Test Plot: In November 1994, test plots were established within the boundaries of the Poen wildfire area and were seeded with various grass and forb cultivars to determine their establishment and long-term success. A small 5-foot-wide Truax drill was used, and the drill was calibrated as closely as possible to the recommended Natural Resources Conservation Service (NRCS) rates for each variety. The degree of cultivar success in these demonstration plots will help determine which varieties will be used for future seedings in the NCA. The following cultivars were seeded:

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[a] Metric conversion: 1 lb/ac = 1,122 g/ha

Cultivar	Variety	Seeding rate	
		(lb/ac pls)	(lb/ac bulk)
Snake River wheatgrass	Secar	8.0	11.5
Cicer milkvetch	Lutana	9.8	12.0
Thickspike wheatgrass	9021076	8.0	7.6
Basin wildrye	Magnar	9.6	10.5
Basin wildrye	Trailhead	9.8	10.5
Bottlebrush squirreltail	SFP-91-UC18	7.3	9.0
Bottlebrush squirreltail	SFP-92-UC17	8.4	9.0
Desert wheatgrass	Nordan	5.7	6.0
Crested wheatgrass	Hycrest	5.7	6.0

Stop 2: *Lepidium* site near Kuna Butte

Ann DeBolt, BLM Botanist, discussed this loamy 7-10" ARTRW/STTH2 ecological site located on the east flank of Kuna Butte. The site had been set aside to determine the effects of drill seeding on slick-spot peppergrass *(Lepidium papilliferum)*, a BLM sensitive species, and its slick-spot habitat. Paired treatments, including a drill seeding and a nonseeded control, were established in a randomized block design and were evaluated for 2 growing seasons. *Lepidium* density, size class, and reproductive output were measured, as was percent cover of all plant species. A rillometer was used to measure mechanical modification of the slick-spot habitat due to the rangeland drill.

The following species were seeded on this site:

Species	Variety	Rate (lb/ac)[2]	Method	Date
Bluebunch wheatgrass	Secar	4.0	rangeland drill	11/96
Siberian wheatgrass	P-27	1.0	rangeland drill	11/96
Standard wheatgrass	Nordan	0.5	rangeland drill	11/96
Wyoming big sagebrush	(collected in Sanpete Co, Utah)	0.16 pls **5.66 lb/ac**	aerial	1/97

Stop 3: Christmas Mountain Control Tower; National Guard Orchard Training Area

Captain Matt Hengel, Idaho Army National Guard (IDARNG) Range Control Officer, discussed the operation of the National Guard's Orchard Training Area (OTA), a 138,000-acre (55,890-ha) military training area on public lands within the NCA, 13 miles south of Boise. The OTA consists of a 58,000-acre (23,490-ha) Impact Area, which contains live firing ranges and target systems used for tank, artillery, and small-arms training purposes. Movable targets are controlled, and tank firing is monitored from the Christmas Mountain control tower. An 80,000-acre (32,400-ha) Maneuver Area surrounds the Impact Area and supports tracked- and wheeled-vehicle maneuver training. Captain Hengel also discussed IDARNG's improved wildfire suppression capability, which has been developed in coordination with BLM's fire management staff.

Steve Knick, USGS Research Wildlife Biologist, discussed the impacts of habitat fragmentation caused by military training. As viewed from Christmas Mountain, the OTA Impact Area is dominated by exotic annuals because of repeated burning caused by firing activities and tracer rounds. Maneuver areas to the north are a mosaic of sagebrush and annual grass types. These sagebrush stands are the most extensive contiguous stands remaining in the NCA, but they provide less productive jackrabbit habitat than stands unaffected by tank maneuvers. Fragmentation of shrublands carries opposing connotations. Numerous small shrubland patches have greater amounts of edge between the shrub patch and surrounding grassland, which facilitates cheatgrass or exotic plant invasion (and subsequent fire spread) into the patch. As a consequence, the probability of losing the smaller patch to fire is greater than for large, undisturbed shrub patches. However, the small patches also can provide a seed source for restoration of shrublands.

Dana Quinney, of the IDARNG Environmental Management Office, provided a general discussion of IDARNG's habitat restoration program, in which they seed selected areas with native species to restore habitat. Some sites have been aerially seeded with Wyoming big sagebrush, but most sites are small enough to have been broadcast seeded by hand.

Stop 4: IDARNG Obsidian Big Sagebrush Restoration Site

Marjorie McHenry and Dana Quinney of the IDARNG Environmental Management Office discussed IDARNG's restoration of a small sagebrush flat northeast of Christmas Mountain. This area was hand-broadcast seeded to Wyoming big sagebrush in 1993. Since then, additional acres have been similarly seeded each year, which will continue until the entire flat is restored. Some of the plants are too small to be easily seen over the annuals, especially those in the southeastern portion of the flat. However, monitoring has shown that the seeding has significantly less cheatgrass than the adjacent unseeded control area.

Snake River Birds of Prey NCA Habitat Restoration Workshop Summary and Recommendations

SNAKE RIVER BIRDS OF PREY NATIONAL CONSERVATION AREA HABITAT RESTORATION WORKSHOP

Boise State University
June 23-25, 1999

Workshop Summary and Recommendations

INTRODUCTION

On June 23-25, 1999, approximately 60 scientists and land managers met in Boise, Idaho, to discuss questions related to the restoration of sagebrush and salt desert shrub habitats in the Snake River Birds of Prey National Conservation Area (NCA). Questions addressed by workshop participants were grouped into three categories – landscape-level restoration, site-specific restoration, and restoration management. Each question is listed below, followed by a consolidated summary of the workshop participants' recommendations. The recommendations constitute the first step in BLM's long-term strategy for restoring the NCA. It should be noted that the recommendations reflect unedited input of workshop participants and have not been evaluated to determine whether they should be incorporated into a Habitat Restoration Plan (HRP) that will guide and coordinate restoration and protection activities throughout the NCA. The recommendations will be reviewed through a systematic analysis by an interdisciplinary team of BLM resource specialists and interagency scientists. Recommendations that are determined to be appropriate will be incorporated into the HRP, which will establish the NCA-wide resource objectives and the management that will be implemented to effect ecological change on a broad scale. Once the HRP is complete, component plans will be prepared for major activities, such as wildfire management, military training, livestock grazing, and recreation, to ensure that these activities are conducted in a manner consistent with the HRP. The recommendations will undergo further analysis as they are incorporated into alternatives considered for future military use of the Orchard Training Area in the National Guard's Environmental Impact Statement and associated decision-making processes.

Since the late 1970s, more than 300,000 acres of shrublands have been lost to wildfires. Large-scale replacement of native shrub and perennial grass habitat by exotic annual grasses and forbs, catalyzed by dramatic increases in the size and frequency of wildfires, is causing significant declines in important prey (black-tailed jackrabbits and Paiute ground squirrels [formerly known as Townsend's ground squirrels]) and raptor (golden eagle and prairie falcon) species. Annual vegetation forms a continuous mat of fine fuels and has changed the natural fire cycle in the NCA from every 50-80 years to about every 5 years. If present trends continue, the area will be completely converted to annual vegetation that cannot support the abundance and diversity of birds of prey that the NCA was established to protect. In addition, communities on the periphery of the NCA face increasing threats from fast-moving wildfires.

For a broader discussion of the resources and programs of the NCA, please refer to the "**Introduction to the Snake River Birds of Prey National Conservation Area**" included in the proceedings of the Sagebrush Steppe Ecosystems Symposium.

LANDSCAPE-LEVEL RESTORATION

1. **Question: Given existing fire frequency and fire suppression capabilities, what is the appropriate spatial design for restoration projects in order to reestablish 1979 vegetation patterns?** [Note: BLM chose to attempt to reestablish 1979 vegetation patterns for 2 reasons: 1) Prior to 1979, few large wildland fires had occurred in the NCA and most of the large shrub stands still existed. Most of the large, devastating fires occurred in the early 1980s and mid 1990s. 2) BLM has very little reliable vegetation-community information dated before 1976.]

   Recommendation: BLM should restore large blocks of sagebrush and/or salt desert shrub habitat, 10,000 to 50,000 acres (4,050 to 20,250 ha) in size. The blocks should be dispersed across the landscape. Habitat mosaics and connectivity should be developed within and between the blocks. When planning the restoration of each habitat block, managers should consider wildlife needs, including the prey base and the raptors that will use the area, especially prairie falcons. Habitat blocks should be planned and designed in such a way that various successional stages are represented and so that existing shrub habitat is incorporated and protected. Also, defensibility against wildfire should be designed into the habitat block, i.e., include some low-hazard fuels in the mosaic.

2. **Question: Given this landscape design strategy, where within the NCA should restoration efforts be focused?**

Recommendation: Where possible, BLM should focus habitat restoration efforts in: 1) areas adjacent to existing sagebrush stands to create the largest possible habitat blocks, 2) sites within 3 miles of the canyon in order to enhance those habitat areas located closest to existing prairie falcon nest sites, and 3) areas experiencing the greatest raptor declines.

Considerations to be used when selecting sites to be restored should include the following (in no particular order):

~ Protect existing native habitat and restoration projects.
~ Give sagebrush sites higher priority than salt desert shrub sites because they exist in higher rainfall zones and, as such, are usually much easier to rehabilitate.
~ Don't ignore shadscale sites; although they don't represent a large area, they provide excellent habitat.
~ Consider raptor use and home ranges.
~ Consider site potential, including precipitation and soils.
~ Strive for connectivity with other habitat.
~ Consider wildfire potential/protection.
~ Use prescribed fire and emergency fire rehabilitation funding opportunities.
~ Incorporate flexibility and time.

3. **Question: How can fuels management strategies be incorporated into restoration projects?**

Recommendation: Fuels management projects, such as fire breaks, greenstrips, etc., are developed primarily to enhance wildfire containment and suppression efforts. As such, they are meant to generally reduce fuel loads and/or flammability and do not normally incorporate habitat-enhancing characteristics. For instance, managers wishing to reduce fuel loads in a particular area might use soil-disturbing practices such as extremely heavy grazing, plowing, or other methods. The level of soil disturbance is usually secondary to the overall fuels management objective. However, workshop participants felt that BLM should strive to incorporate habitat-enhancing features in these types of projects whenever possible.

BLM should minimize soil disturbance to reduce cheatgrass infestations and should emphasize maintenance of existing perennial vegetation. Projects should be designed to include natural fuel breaks. This would include maintaining a mosaic of shrubs and perennial grasses as well as incorporating discontinuous fuels and existing fuel/fire barriers (roads, rock outcrops, etc.) into the project.

Initial habitat restoration planning should be done on a landscape basis, but managers should set site-specific project goals before and during each restoration project. The goals should incorporate the following considerations:

~ fire suppression/protection
~ fuels management
~ fuel breaks, particularly along major roads, railroads, and other areas of permanent disturbance
    - breaks should be at least 300 feet wide
    - treatment may include herbicides, mechanical techniques, intensive livestock grazing (requires either more fences or active livestock handling techniques), and prescribed fire
    - greenstrips are preferred over mechanically disturbed areas such as plowed firebreaks
    - design should consider access, prevailing winds, ignition sources, and preexisting barriers
~ broadcast use of herbicides over large areas to reduce annual weed infestations
~ possible exclusion of livestock from specific sites to enhance development of biological soil crusts and to retard cheatgrass infestation

4. **Question: What type of remote sensing technology is appropriate for use in planning a landscape-level restoration strategy?**

Recommendation: Satellite multi-spectral scanning (MSS) imagery that provides fine-scale information is very useful. Managers should also use aerial photography of whatever scale and resolution is appropriate for the particular project.

SITE-SPECIFIC RESTORATION

1. **Question: What plant species should be used to restore prey habitat in the various soil/precipitation zones of the NCA?**

Recommendation: When considering which shrub, forb, and grass species to use in a particular habitat restoration project, managers should first determine the objectives of the project. This means determining whether the project is being planned for habitat restoration, emergency fire rehabilitation, or fuel break purposes. Once this determination is made, managers may want to develop a decision tree that shows which species to use in which situation. Considerations to be used in developing the decision tree could include:

- ecological site capability
- adaptability of species to site
- seed availability
- likelihood of success (life forms/structure)
- rooting depth

Potential species to be seeded in either sagebrush or salt desert shrub sites include the following:

Sagebrush Sites

Native
Wyoming big sagebrush
Thurber needlegrass
Basin wildrye
Sandberg bluegrass
Squirreltail
Rabbitbrush
Bluebunch wheatgrass
Thickspike wheatgrass (off-site native)
Silver sage (off-site native)
Spiny Hopsage
Horsebrush
Lupine spp.
Indian ricegrass
Sand dropseed
Western wheatgrass
Biological soil crusts

Non-Native
Russian wildrye
Crested wheatgrass
Siberian wheatgrass
Prostrate (forage) kochia

Salt Desert Shrub Sites

Native
Indian ricegrass
Winterfat
Shadscale
Sand dropseed
Budsage
Spiny hopsage
Needle-and-thread
Fourwing saltbush
Sandberg bluegrass
Squirreltail
Nuttall's saltbush
Greasewood
Saltgrass
Biological soil crusts
Lupine spp.
Crepis spp.
Rabbitbrush

Non-Native
Crested wheatgrass
Russian wildrye
Prostrate (forage) kochia
Siberian wheatgrass

Specific recommendations included the following:

- seed crested wheatgrass at no more than 3 lb. per acre
- encourage the commercial seed development of more site-adapted forbs
- use forage kochia only for greenstrips and fuels management projects

2. **Question: How can we improve availability of native seed?**

Recommendation: Managers should identify their basic needs for specific species over a defined time period to ensure a market for growers. Considerations to be used in determining basic needs include:

- Base determination on historic low-year needs to increase likelihood that all seed will be used.
- Combine interagency needs for a specific species within a region if possible.
- Develop improved funding mechanisms, which may include payment in advance.
- Consider subsidizing growers in the beginning in order to develop certain seeds in commercial quantities.
- Consider the need for government agencies to do basic research to improve seed production.

Managers can improve the availability of seed by using forward contracting and regional warehousing of identified germplasm. Forward contracting simply means contracting for a crop which is yet to be harvested or even yet to be grown. This gives seed producers a specific length of time to develop a seed source and guarantees them a market for the developed seed. Coordination should be improved with seed producers, and seed production should be contracted with multiple producers to optimize production and quality. To avoid the high cost of critical seed during mid-summer shortages, seed should be purchased in the off-season when prices are low. Managers should identify and manage sites for wildland collection and maintenance of seeds (germplasm sources).

3. **Question: Are there better methods available to control competition from exotic annuals? If so, what?**

Recommendation: Managers should first manage sites to restore natives, while protecting residual perennial understory where it exists. Managers can use sequential plantings in restoration projects. This means species that suppress exotics are seeded first. These species are subsequently removed, and native species are seeded in their place. Native species also could be overseeded directly into the existing stand.

BLM should use herbicides and changes in livestock use in a cooperative effort to restore shrub communities. Livestock season of use should be changed to enhance perennials and biological crusts through flexible grazing dates, adaptive management, and removal of livestock based on perennial growth and cheatgrass maturity.

Other methods of cheatgrass suppression, such as biological controls, should be investigated. For example, BLM should keep current on the latest cheatgrass smut research to determine if and when it may be suitable and cost-effective for broad-scale cheatgrass control.

4. **Question: What additional technologies could be used to enhance restoration success, particularly for native species?**

Recommendation: A number of technologies exist to enhance the success of restoration projects. They include the following:

Equipment/technology: Use equipment and techniques that minimize soil disturbance and enhance water retention, such as no-till drilling and light chaining/harrowing. Also, improve seeding efficiency and versatility by using equipment that will allow for seeding multiple species at the same time.

Shrub thinning in decadent stands: Use thinning techniques that minimize soil disturbance, such as chemical application and winter burning. Community diversity could be improved by interseeding with desirable perennials after thinning.

Seeding: Use seeding to increase/improve long-term diversity by striving for succession rather than a 1-time application. This may entail overseeding into a depauperate native stand. Managers also should work toward better seed production, cleaning, and storage techniques.

Site preparation: Use carbon amendments to control nitrogen levels. Also, monitor chemical and biological soil data, such as K sequesters and soil biota to enhance seeding success.

Weather data: The Agricultural Research Service has established several weather-monitoring stations around the NCA and is using data collected to predict future weather patterns. These data should be used when planning restoration projects.

5. **Question: What are the short-term effects of restoration projects on key wildlife species? If those effects are negative, how can they be minimized?**

   Recommendation: The potential effects of restoration projects on wildlife species include displacement and direct/indirect mortality. Managers should not focus too narrowly on a few key species and ignore the larger food web. Tilling causes organic matter to decompose faster, which may adversely affect certain species that rely on this organic matter.

   Potential mitigation for some of these effects include the following:

   - Stagger restoration activity to reduce landscape-scale impacts.
   - Enhance food for ground squirrels by applying "OUST®" at rates that control cheatgrass but not Sandberg bluegrass.
   - In special cases and in limited areas, plant and/or water seedlings or seedings.

6. **Question: How can we conduct restoration projects while protecting and maintaining sensitive slick-spot peppergrass (*Lepidium papilliferum*) populations?**

   Recommendation: Managers should conduct good inventories and use accurate maps showing sensitive species' locations. The inventories and maps should be updated prior to management actions such as herbicide applications or forage kochia seeding. Managers should strive to maintain perennial vegetation and to restore sagebrush as soon as possible. Conduct drill-seeding operations later in the year to allow for native-seed dispersal. Seeding methods that minimize surface disturbance (i.e., no-till drilling) should be considered over the traditional rangeland drill. Minimize surface disturbance by using refined tools, such as a no-till drill. Block out *Lepidium* from seeding operations by fencing or by covering with tarps. Use more specific herbicides (Post/Fusillade) that have less impact on *Lepidium* or consider using non-chemical methods, such as carbohydrate amendments, to control cheatgrass. Use plant materials that won't compete with *Lepidium.*

7. **Question: What additional information or research is needed to better plan and implement a restoration strategy on a landscape basis as well as specific restoration projects?**

   Recommendation: The following research suggestions are listed in priority order. Parenthetical numbers are the result of 3 votes per participant.

   1. Rearing and harvesting of native species (25)
   2. Equipment (multi-species, durable, vary depths, irregular seeds, light-weight, no-till drill) (20)
   3. Smut dusting and other biological controls (20)
   4. Habitat patch size (max/min), connectivity, and distribution, considering raptors, prey, fire (20)
   5. Ecological outcomes of seed mixes, seeding rates (19)
   6. Greater understanding of factors that affect restoration success (19)
   7. Restoration of salt-desert shrub (19)
   8. Biological soil crust restoration – how, which, where? (18)
   9. Research nutrient cycling; reverse fertilization (15)
   10. Effect of OUST® on other vegetation and various mammal species (9)
   11. Forage use by ground squirrels, jackrabbits, grasshoppers, etc. (9)
   12. Research of seed germination requirements, particularly of forbs (8)
   13. Remote platform for vegetation monitoring (8)
   14. Animal response to restoration projects (7)
   15. Thresholds of development for invasion of exotics (6)
   16. Microbial components of ecosystem (6)
   17. Integration of weed control methods (6)
   18. Firebreak width, placement, and vegetation types (5)
   19. How to restore *Lepidium* sites – effects of chemicals, reintroduction into restored sites (5)
   20. Effects of herbicides on non-target species (5)
   21. Ground-truthing of remote-sensing models (4)
   22. Role of soil biota in seed successes and failures (4)
   23. Alternatives to species replacement; invasiveness of the alternatives (4)
   24. Better knowledge of seasonal weather patterns (3)
   25. Seed storage for natives with short-term viability (3)
   26. Determine cheatgrass invasion/crust disturbance threshold – treatment or all-out war (3)
   27. Use of aerial photography (2)

28. Long-term weather forecasting (2)
29. Climate and fire behavior – future changes (2)
30. More info about additional chemicals (2)
31. Satellite MSS and fine-scale info (1)
32. Ways to alter soil chemistry to enhance seeding success (1)
33. Effects of management actions on squirrels and other ecosystem components (1)
34. Monitor vegetation biomass composition and turnover (1)
35. Confront temporal/spatial question, variation over space and time (1)

RESTORATION MANAGEMENT

1. **Question: How should uses such as livestock grazing, military training, and recreation be adjusted to maintain restored landscapes? (How do we protect our investment?)**

   Recommendation:

   Grazing: Concentrate grazing use in winter to reduce impacts on perennial vegetation or institute grazing systems where appropriate. Keep artificial water sources out of sensitive areas (sagebrush and *Lepidium* sites), and exclude grazing from critical areas. Provide for rest from grazing after a restoration treatment. The length of the rest period should be based on the needs of the vegetation, not a specific time period.

   Military training: Modify the season of use for some military activities to reduce impacts to soils and seedlings of restored landscapes. Continue on-going training restrictions in shrub communities and critical areas to minimize ecological damage, including restricting tanks from maneuvering on wet soils and restricting firing activities during times of extreme fire danger. Also, if transportation of weeds becomes a problem, ensure that military vehicles coming from outside the area are washed before they enter the Orchard Training Area (OTA).

   Recreation: Complete the road-designation process for the entire NCA. Reduce the number of recreational access roads. Close non-designated areas. Determine seasons of use for open areas. Determine if transportation of weed seeds by recreationists and other users is a problem, and address if needed.

   General: Set short-term objectives for restoration. Exclude soil-disturbing activities such as grazing, military use, and recreation until plant community objectives are reached. Set and monitor long-term landscape/management objectives, and review the restoration process in context of the need for continued fuels management. Also, develop a strong education plan for NCA users about the public benefits of habitat restoration.

2. **Question: What criteria should be used in deciding whether to change management of existing uses or to actively restore desired vegetation through weed control and seeding in order to reach restoration objectives?**

   Recommendation: First, determine whether any stands are in a condition that could be improved by management alone. Considerations to be used in this determination include the following:

   ~ Status of understory vegetation (perennial vs. annual composition)
   ~ Existing shrub cover – is it greater than 15%?
   ~ Density of perennial grasses, exotics, annuals, forbs
   ~ The presence and condition of biological crusts
   ~ Whether the site has been previously reseeded.
   ~ Potential for encroachment of other invasive weeds
   ~ Wildfire potential – fuel density; standing dead/dormant organic matter

   Second, determine the potential for site improvement/restoration. For instance, rather than doing a full-blown restoration, herbicide might be used to release remaining perennials in existing shrub stands.

   Proposed management changes must have high probability of both success and benefit to attract users as partners.

   Site location is also an important attribute to consider. Is the site remote enough that it will not need to be protected from human or livestock use? Also, what about access for fire suppression?

3. **Question: How do we successfully integrate the Emergency Fire Rehabilitation (EFR) program into the restoration strategy?**

   Recommendation: BLM should review the existing EFR program and make it more flexible. EFR funding should be extended to 3 years to ensure adequate funds for reseeding if the original EFR seeding is unsuccessful. EFR funding should allow for control of exotic annuals with chemical or biological tools; without this control, the entire EFR project is often jeopardized.

   The USDI and USDA should create a common restoration funding authority that would integrate EFR fuels management with other funding. This would allow BLM's hazardous fuels management program to fund all aspects of fuels management, including seeding of fire-resistant vegetation.

   Recommendations specific to the BLM Lower Snake River District include:

   - Use forage kochia only in greenstrips and fuels management projects.[a]
   - Use EFR to implement the NCA's road management plan by closing and rehabilitating unneeded roads during EFR project implementation.
   - Identify preferred plant materials for EFR projects in the Habitat Restoration Plan.
   - Obtain additional funding; give restoration priority to areas adjacent to existing shrub communities.
   - Restore previously unsuccessful EFRs through a long-term integration with future EFR and habitat restoration projects.
   - Develop a rating criteria in the Habitat Restoration Plan to determine if, in a given year, EFR sites are priorities when compared to planned restoration sites.

4. **Question: How do we measure success for site-specific restoration projects? Evaluation/Monitoring?**

   Recommendation: Managers must set management goals and ensure those goals are being met through pre- and post-restoration monitoring. The monitoring should be both short-term (first 3 years), to determine the success of the seeding, and long-term (5-year intervals), to determine any measurable changes in shrub cover. Monitoring should measure the response of specific habitat features, such as:

   - Vegetation composition, density, trend, species arrangement, reproduction/recruitment, and weed composition
   - Relative abundance of small mammal, reptile, insect, and passerine species
   - Prey presence and abundance (in areas with and without plant community)
   - Raptor abundance and reproduction
   - Soil food webs
   - Forage use by small mammals, insects, and livestock

   When interpreting monitoring results, managers should consider physical features, such as climate and soils. Results should be compared against a standard, such as native sites in good condition.

   When monitoring, the following site characteristics should be measured:

   - Biological soil crust – cover, diversity
   - Weeds – cover, diversity
   - Vegetation – cover, diversity
   - Recruitment – size classes
   - Soils – physical, chemical, biological properties

5. **Question: How do we determine if landscape-level goals are being achieved?**

   Recommendation: To measure success, managers must define "success" for all trophic levels. The success of landscape-level goals may be monitored in the NCA through the following:

   - Measure trends in abundance and productivity of nesting raptors (primary: golden eagle and prairie falcon; secondary: ferruginous hawk and burrowing owl)
     - Canyon nesters (biannual full canyon surveys)
     - Benchland nesters (quadrants)

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[a] This recommendation resulted from concerns expressed by numerous workshop participants that not enough was yet known about the effects of forage kochia on native vegetation. Although they recognized the potential of the species for use in greenstrips and fuels management areas, most participants felt that BLM should restrict its use to areas where native species were less important, at least until more is known about the species.

- Measure trends in relative abundance of primary animal species throughout the NCA
    - ~ Jackrabbits (spotlight transects – traditional routes – for 2-3 yrs. every 10 yrs. to determine population cycles)
    - ~ Ground squirrels (trapping indexes – annually)
    - ~ Non-prey species (every 5 yrs.)
- Compare actual with desired plant communities
    - ~ Develop a GIS layer showing presence/absence of weed species
    - ~ Use remote sensing to measure shrub patch size, distribution, and connectivity (every 5 years)
- Determine if communities are self-perpetuating or shifting over time
    - ~ Use satellite imagery at 5-10 year intervals
    - ~ Map site-specific monitoring results
- Measure size and frequency of fires
    - ~ Use annual reports
    - ~ Summarize at 5-year intervals
- Use remote sensing combined with site monitoring
    - ~ Use control plots
    - ~ Develop study design to determine if changes result from management action

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Boise State University
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